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"They Are Drying Out": Social-Ecological Consequences of Glacier Recession on Mountain Peatlands in Huascarán National Park, Peru

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“They Are Drying Out”: Social-Ecological Consequences of Glacier Recession on Mountain Peatlands in Huascarán National Park, Peru

by

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Dissertation

Presented to the Faculty of the Graduate School of

The University of Texas at Austin

in Partial Fulfillment

of the Requirements

for the Degree of

Doctor of Philosophy

The University of Texas at Austin

May 2016

Acknowledgements

The journey to the Cordillera Blanca and Huascarán National Park started with a whiff of tear gas during the *Arequipazo* of 2002. During a field season in Arequipa, the city erupted into civil unrest. In a stroke of neoliberalism, newly elected President Alejandro Toledo privatized Egesur and Egasa, regional power generation companies, by selling them to a Belgian entity. Arequipeños violently protested President Toledo's reversal on his campaign pledge not to privatize industries. Overnight, the streets filled with angry protesters demonstrating against "los Yanquis." The President declared a state of emergency that kept Agnes Wommack and me trapped in our hotel because the streets were unsafe for a couple of Yanquis. After 10 days of being confined, we decided to venture out to the local bakery. Not long after our first sips of coffee, tear gas floated onto the patio and the lovely owner rushed out with wet towels to cover our faces. With that, I knew the field season was over.

As soon as the airport reopened, Agnes returned to the U.S. Meanwhile, my advisor Kenneth R. Young suggested that I go to Huaraz to help a fellow graduate student, Jennifer Lipton, with her field work. I finally made my way back to Lima and then to Huaraz. The crisp mountain air and tranquility in Huaraz were a relief. Over the next week, I tagged along with Jennifer, collecting GPS points. I fell in love with the dramatic, ice-covered jagged peaks of the Cordillera Blanca and knew that I would someday return. What I could not have known at the time is that it would take 7 years and a venture into commercial real estate before I would return.

It has been said before and is worth repeating here: a dissertation appears as a sole-authored product, but the truth is that a huge cast of characters makes it possible. First and foremost, I thank my advisor, Kenneth R. Young, for offering me the incredible opportunity to be his Graduate Research Assistant on a 3-year grant from the National Science Foundation's Coupled Natural Human Systems program (CNH: Collaborative Research: Hydrologic Transformation and Human Resilience to Climate Change in the Peruvian Andes" Award Number 1010550 with Co-PIs Jeff Bury, Bryan G. Mark, and Mark Carey). After several years of being out of touch, Ken and I met for coffee on a September afternoon, only days after he received news that the team had been awarded the prestigious CNH grant. He offered me the position and I was ready to start immediately. The serendipity of that meeting changed the course of my life.

Ken has the knack for balancing gentle guidance with the ability to tighten the screws at the right moment. Along the way he gave me enough clues to be able to piece together the resulting research design, data collection, analysis, and conclusion that you'll read here. His fingerprints are scattered throughout and I'm honored to have collaborated so closely with him. While in Peru, Ken and Blanca León were like family, including me in breakfasts, lunches, dinners and making sure I was comfortable a long way from home. We spent many hours together on our hands and knees counting tiny plants in Huascarán National Park and these were some of the most joyful times of the project. Our celebratory dinners at Encuentro were always memorable.

Another key partner in the project is Asunción Cano Echevarria, who patiently taught me the plant species over two seasons. Always cheerful under any conditions, Asunción is an outstanding person to work with in the field. I am especially grateful for his interest, dedication, and contributions to my project. Chapter 5 wouldn't have been possible without his help. I am also grateful to Maria Isabel de la Torre for her friendship and for refining the grass identifications.

My dissertation committee has been unwaveringly supportive along the entire process in many ways. Kelley A. Crews has inspired me towards excellence and motivated me to reach further than I thought was possible – this has been true since we met in 2001. Gregory W. Knapp gamely tolerated my systems framework. His critical eye towards systems helped me think carefully about the limitations of such frameworks. Francisco L. Pérez's meticulousness, elegance, attention to detail, and enthusiasm for mountains are a source of constant inspiration. His reminders to enjoy life and prioritize my values arrived at the perfect time. Daene C. McKinney made one comment during my dissertation proposal presentation that sent me on the quest to find a regression model that could predict changes in wetland area. He also made numerous helpful introductions in Huaraz and invited me to participate meaningfully in a multinational conference and global south-to-south technological exchange regarding hazards in high mountains. Although he wasn't formally on my committee, Eugenio Arima opened my eyes in important new ways. It was through his courses that I gained the insight to connect land change science and econometrics.

In Huaraz, I owe a special thanks to Ing. Jesús Gómez López, the director of Huascarán National Park, for granting permission to conduct research in the park (SERNANP Permit TUPA N^{o} PNH-008-2012), for supporting the many scientists who work there, and for his unflagging dedication to park sustainability. At The Mountain Institute, Jorge Recharte, Beatriz Fuentealba, and Laura Trejo have been partners in the project and I thank them for their dedication to protecting landscapes and cultures in the Cordillera Blanca. Zarela Zamora offered a home away from home over four field seasons. I can personally attest to her legendary hospitality and resourcefulness and am lucky to count her as a friend. Zarela introduced me to Gladys Jimenez, who was instrumental in the interviews with local people. Gladys instantly understood the project and took a personal interest in it. Her knowledge of local customs and culture fine-tuned the interviews to capture the richness and texture expressed in both Spanish and Quechua. I am also grateful to Bruno Revirón, our tireless guide, outfitter, and restauranteur, who put me together with Abraham Aldave. Abraham kept me well-nourished at high altitudes and scared off the pumas by lighting matches in the dark of night.

The support I received from fellow graduate students at UT was phenomenal. I am especially grateful for help from Bisola Falola, Paul Holloway, Naya Jones, Lindsey Carte, Kalli Doubleday, Kaiti Tasker, Aaron Groth, Anwar Sounny-Slitine, and Xuebin Yang. From the Digital Landscape Lab and the Spatial Science Lab, I thank Mario Cardozo, Niti B. Mishra, Thoralf Meyer, and Thomas Christensen. Special thanks to Julio Postigo and students in his graduate seminar at Pontifica Universidad Católica del Peru who helped me focus and refine the PPD model in the weeks leading up to my dissertation defense. Julio also deserves special credit for pointedly asking me only two weeks after starting the graduate program if I was going to use social-ecological system theory. Laura Read has been a sounding board, comic relief, and responsible for introducing me to all the bakeries in Huaraz that sell *borrachitos* (rum cake bites) that I still crave and can't find in the U.S.

Financial support was provided by numerous institutions internal and external to the University of Texas at Austin. Without these awards, the work would not have been possible. In total, I was awarded approximately $54,000 from the following sources: University of Texas at Austin Graduate School Continuing Fellowship ($29,947 including tuition and fees); DigitalGlobe Foundation Imagery Grant (~$4100); National Science Foundation, Doctoral Dissertation Research Improvement Grant (PI K.R. Young, Grant No. BCS-1333141, Title: Glacier Recession and Implications for High-Elevation Peatlands, $15,881); American Association of University Women, Austin Branch Dissertation Fellowship ($2500); Association of American Geographers, Mountain Geography Specialty Group, Chimborazo Student Research Grant ($500); University of Texas at Austin, Department of Geography and the Environment, Robert E. Veselka Graduate Research Travel Fellowship ($650).

The Transdisciplinary Andean Research Network (TARN) provided an instant network of professors and graduate students in the Cordillera Blanca. It was a distinct pleasure to be associated with Bryan Mark, Michel Baraer, Jeff McKenzie, Mark Carey, Adam French, and Jeff Bury. Having access to the depth of knowledge that these scholars possess was unparalleled. Being with them in the field was always a crash course in glacier dynamics, glacier hydrogeology, and the social and political context. I am especially appreciative of Bryan Mark and Michel Baraer, who generously shared the historic discharge dataset that is the primary input in the model I developed in Chapter 4.

Other researchers that I collaborated with are John Hribljan and Rod Chimner from Michigan Technological University. Seeing the peat cores emerge for the first time was breathtaking and I anxiously await John's analysis.

Finally, I thank my parents, Mary and Jamie Polk, and my sister and brother-in-law, Annie Polk and Chris Oatey, for their unmitigated encouragement, curiosity, and enthusiasm. My first nephew, Jules Polk Oatey, was born during this project in 2012. If he visits Huascarán National Park during his lifetime, I hope that the icy peaks remain and the peatlands are still quaking and spongy. Lastly and mostly, I thank Agnes Wommack, my wife. Her patience, understanding, tolerance, sense of adventure, and emotional and financial support made everything possible, fulfilling, and more fun.

“They Are Drying Out”: Social-Ecological Consequences of Glacier Recession on Mountain Peatlands in Huascarán National Park, Peru

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The University of Texas at Austin, 2016

Supervisor: Kenneth R. Young

Ecosystems that are proximal to tropical glaciers, such as mountain peatlands in Peru’s Huascarán National Park, are experiencing ecological changes caused by glacier recession. Peatlands are not only affected by biophysical changes, but also by human behaviors and associated land use decisions that are influenced by changing environmental conditions. Thus mountain peatlands in Huascarán National Park are coupled natural human systems. Social-ecological system theory can be used to situate integrated research questions and methodologies addressing the transformations of coupled natural human systems. In this dissertation, a social-ecological systems framework called the Press-Pulse Dynamics model was applied to investigate peatland transformations related to climate change and land use decisions. Using social-ecological systems theory necessitates methodological pluralism because reliance on a single epistemology produces limited new knowledge. In this research, a set of quantitative and qualitative techniques were implemented to produce an integrated perspective on social-ecological peatlands. A 23-year spatio-temporal remote sensing analysis of peatlands showed that the ecosystems are losing area through processes of fragmentation, attrition, and isolation. Statistical evidence suggested that loss of glacier area and decreasing stream discharge are driving factors for peatland area change, but that in the future, precipitation may become a more dominant factor. Extensive *in situ* plant surveys indicated high alpha and beta diversity and that there are likely many more species to be added to the known peatland flora. Vegetation heterogeneity was explained by the abiotic factors of elevation, percent organic matter, bulk density, and cation exchange capacity. A series of ecological oral histories showed that local users of peatlands have observed spatial and ecological changes over their lifetimes. The observations by local people corroborated quantitative findings and substantiated the linked biophysical and social aspects that affect peatlands. Interconnections between social and biophysical processes in the PPD model suggest that future peatland stewardship responsibilities should be entrusted to local communities in close collaboration with the national park and non-governmental organizations who could provide technology transfer support. The dissertation contributes to and advances geography by innovatively bridging multiple perspectives through a social-ecological systems model to produce new biophysical and social knowledge about tropical mountain peatlands that are affected by glacier recession.

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Chapter 1: Introduction

Glacier recession is one of the most visually striking manifestations of climate change. Ice loss is tangible and can evoke powerful reactions from observers because the connections between the melting process and a warming planet are intuitive. Melting ice narratives have cast glaciers as endangered species that are struggling to survive and need to be saved (Carey 2007). Like threatened iconic and charismatic megafauna that pull on the heartstrings of the public, charismatic glaciers receive media attention and attract large international research teams to study causes and consequences of accelerating ice loss. New eco-tourism marketing strategies promote climate change tours, such as *La Ruta de Cambio Climático* in Huaraz, Peru, and offer up-close views of melting ice and expanding glacier lakes (Figure 1.1). While charismatic glaciers melt and attract researchers, the media, and tourists, proximal ecosystems that are affected by ice loss have received scant interest. The cryosphere and contiguous ecosystems are sensitive to climatic shifts and are undergoing rapid transformations with new no-analog states emerging (Young 2015). Glaciated tropical mountains such as Peru's Cordillera Blanca are relatively well-studied by glaciologists and other related researchers specializing in the cryosphere. In a complementary way, the changes occurring offer an opportunity to not only investigate the social and ecological consequences of glacier recession, but also the opportunity to pose new and innovative research questions contributing to the human dimensions of global change.

Figure 1.1. Glacier recession in Huascarán National Park. Visitors on the *La Ruta de Cámbio Climatica* tour visit Pastoruri, a glacier that recently split into two small sections.

Glaciers can be found at the highest elevations of tropical mountains. Below the icy features, tropical mountain landscapes are a patchwork mosaic where high altitude peatland ecosystems exist. Peatlands are hybrid terrestrial - aquatic systems where the accumulation of organic matter exceeds the decomposition rate. Extensive examples of high altitude peatlands can be found in Peru’s Huascarán National Park (HNP), where they are commonly called *bofedales*. Peatlands in HNP are likely to be affected by climate-induced glacier recession and anthropogenic disturbances, namely grazing of non-native livestock. A summary of the principal characteristics of HNP peatlands can be found in Table 1.1. Occurring in flat to gentle slopes in valley bottoms, peatlands are usually flat and may contain hummocks. Hydrological inputs are varied and the water table position fluctuates depending on seasonality, with higher water levels during the wet season and lower levels during the dry season. Plant communities are adapted to variable water levels and saturated soils and include cushion plants, sedges, rushes, grasses, asters and Sphagnum moss only occasionally. Soils tend toward high organic matter content and high acidity. Several abiotic and edaphic factors explain the organization of plant communities. Both plants and soils are subject to moderate to high livestock impacts from grazing and trampling by cattle, sheep, and horses.

The focus of this dissertation is on peatlands in HNP as viewed through the lens of social-ecological system theory. The research is motivated by a desire to apply science to managing an ecosystem that is experiencing climate-related transitions. The work aims to inform national park management and other land use decision-makers who lead conservation planning efforts for HNP. To meet this goal and to fill in gaps about tropical mountain peatlands affected by receding charismatic glaciers, the dissertation is organized by three questions:

- How are peatlands changing spatially and what variables drive these changes?
- Which environmental factors account for peatland vegetation heterogeneity?
- Local communities impact and are impacted by mountain peatlands so how do they perceive peatland changes?

To address these questions, I designed an innovative mixed method approach built on extensive *in situ* data collection, geospatial techniques (GIS, remote sensing, and spatial statistics), multivariate ecological analysis and semi-structured interviews. The three research questions and their corresponding methods are situated within a specific SES model premised on the notion that peatlands in HNP are a coupled natural-human system. By implementing an SES model using three questions, the integrative geographic research expands on approaches and topics addressing human-environment interactions. By testing and expanding on an SES model, the research reveals a richly textured quantitative and qualitative spatial analysis about tropical mountain peatlands affected by climate change.

Table 1.1. Typical characteristics of peatlands (*bofedales*) in Peru's Huascarán National Park based on this dissertation research adapted from Squeo et al (2006). Complete vascular plant taxa can be found in Appendix 1.

Characteristic	**Description**
Topographic setting	Situated in valley bottoms on flat to gentle slopes; soligenous (sloped) peatlands also present
Peatland landform	Usually flat with a raised center; hummocks sometimes present
Hydrology	Groundwater, glacier meltwater, sheetflow and precipitation during the wet season, some stream/river influence
Water table position	At or slightly below the surface during the dry season and above the surface during the wet season
Dominant vegetation	Dominated by Poaceae, Asteraceae, Cyperaceae, and other species; *Sphagnum* sp. is absent in quadrats, but was observed in small (~ 1 m^2) isolated patches; refer to Appendix 1 for vascular plant taxa
Plant communities	Cushion plants, sedges and rushes, sedges and grasses, asters and sedges
Soil composition	Poorly decomposed Juncaceae peat with organic matter >46% on average
Soil pH	Acidic, ranging from 3.6 – 5.4
Livestock impact	Moderate to high; large herds comprised of horses, cattle, sheep. Native camelids absent, but some llamas used for tourism.
Abiotic and edaphic factors	Elevation, % organic matter, cation exchange capacity, bulk density

GEOGRAPHIC THOUGHT AND SOCIAL-ECOLOGICAL SYSTEM THEORY

One of the defining themes of geographic research is human-environment interactions (De Blij 2012), a line of inquiry that can be traced back to the influential work of George P. Marsh (Liverman, Yarnal, and Turner II 2003). As early as 1864, Marsh observed and documented the role of human agency on the landscape. He remarked that "the effects of human action on the forms of the earth's surface could not always be distinguished from those resulting from geological causes (Marsh 1965, 48). Marsh's detailed observations inspired Carl Sauer, William L. Thomas, Billie Lee Turner II, and many others to evaluate the role of human agency in shaping the landscape from various integrative perspectives that join natural and human systems, or biophysical and social processes (Liverman, Yarnal, and Turner II 2003). Of particular interest to geographers are landscape patterns and processes that affect and are affected by biophysical and social systems and the feedbacks between them (Liu et al. 2007; Turner II 2010). Because these systems interact in complex ways and across local, regional, and global scales, interdisciplinary approaches are advocated in order to better understand these so-called coupled natural human systems (Baerwald 2010; National Research Council 2010).

A paradigm closely related to coupled natural human systems is social-ecological system (SES) theory (Young et al. 2006; Turner II 2010), a framework that seeks to understand human dependence on natural resources and ecosystem services and how ecosystem dynamics influence and are influenced by human activities (Chapin, Folke, and Kofinas 2009). Much like the coupled natural human system paradigm, SES theory presumes that humans and the environment are inextricably linked. SES theory embraces change as a basic feature, implying that humans should manage the environment in a manner that fosters resilience to change (Chapin, Folke, and Kofinas 2009).

In 2011, Collins and colleagues (2011) published a new representation of SES theory called Press-Pulse Dynamics (PPD) (Figure 1.2). The goal of the PPD model is to understand the connections and feedbacks between biophysical and social systems by analyzing disturbances and related impacts on ecosystem services. Given that change is a key feature in an SES, it characterizes a system based on two disturbances, press or pulse. The disturbances are differentiated by temporal qualities. A press is chronic, where many disturbances occur frequently and constantly, whereas a pulse is a discrete change with few disturbances occurring infrequently (Smith, Knapp, and Collins 2009). Linking the biophysical and social templates are disturbances and ecosystem services. Connections within the system are represented by an arrow and can be tested with one or more hypotheses. For example, every arrow in Figure 1.2 represents a possible hypothesis or set of hypotheses. The model is designed to be the foundation for long-term ecological transdisciplinary research across geographic and institutional scales. It also presumes that collaboration and interdisciplinary methodologies should enhance the ability to understand complexity. The framework has many advantages: it is innovative, it operates on multiple scales, it is sensitive to local needs and values diverse ways of knowing, it seeks input from all stakeholders, is inclusionary, and endorses local control, and was designed for long-term ecological research. Finally, it is iterative and hypothesis driven (Collins et al. 2011).

For much of the 20th century, disciplinary boundaries generally isolated social scientists and ecological scientists from one another. Broadly speaking, social scientists paid little attention to the role of ecosystems and ecologists excluded humans from their work. But after the 1908s, several subfields emerged that bridged social and ecological sciences and propelled scholars to shift their perspectives towards coupled natural human systems (Berkes, Colding, and Folke 2003). According to Berkes and colleagues (2003), SES frameworks originate in the six uniting subfields: environmental ethics, political ecology, environmental history, ecological economics, common property, and traditional ecological knowledge. The reader is directed to Berkes et al. (2003) for descriptions and examples from each of these subfields.

In addition to the subfields noted by Berkes et al. (2003), ecologists are responsibile for developing SES frameworks, including the PPD model. The United States National Science Foundation established the Long Term Ecological Research (LTER) network in 1980, a program that makes decadal scale observations and experiments in multiple ecosystems across North America. The mission of the LTER is to generate “the knowledge and predictive understanding necessary to conserve, protect, and manage the nation's ecosystems, their biodiversity, and the services they provide” (http://www.lternet.edu/network/). While working on integrated social and ecological systems, LTER researchers – led by Scott Collins – recognized an opportunity to develop a model that could be supported by the existing LTER network infrastructure (Robertson et al. 2012). From this extensive constellation of research sites and a long legacy of scientific results, the Press Pulse Dynamics (PPD) model emerged as a guide for long-term research that integrates social science and ecology. The LTER network adopted the PPD model in order to implement standardized terminology across disciplines and to facilitate the evaluation of SESs (Collins et al. 2011). While there are numerous published SES frameworks available, the PPD model was chosen for this dissertation for its central emphasis on spatial and ecological change, disturbances, and human perceptions of change, characteristics that play important roles in Huascarán National Park.

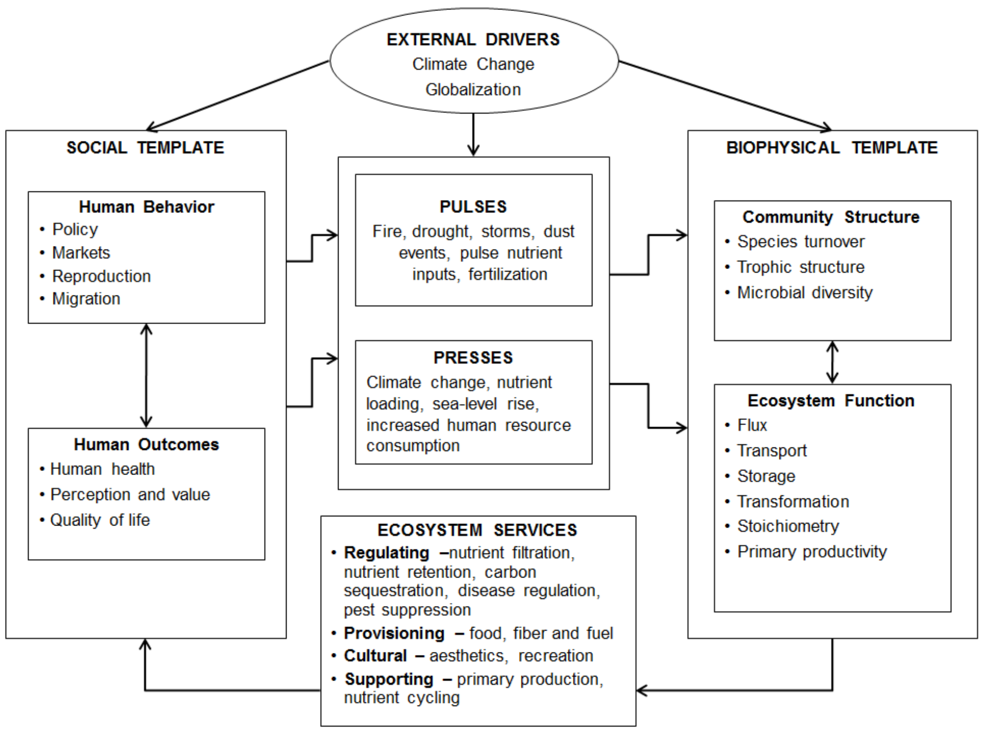


Figure 1.2. Press-Pulse Dynamics framework from Collins et al. (2011).

Huascarán National Park Peatlands and the PPD Model

SES theory and the PPD model are potentially useful frameworks for understanding the links between biophysical processes, ecosystem services, and perceptions of environmental change as they relate to changing peatlands in HNP. In this section, I explain why this could be and how each component in the PPD model could be applied to peatlands in HNP as illustrated in Figure 1.3.

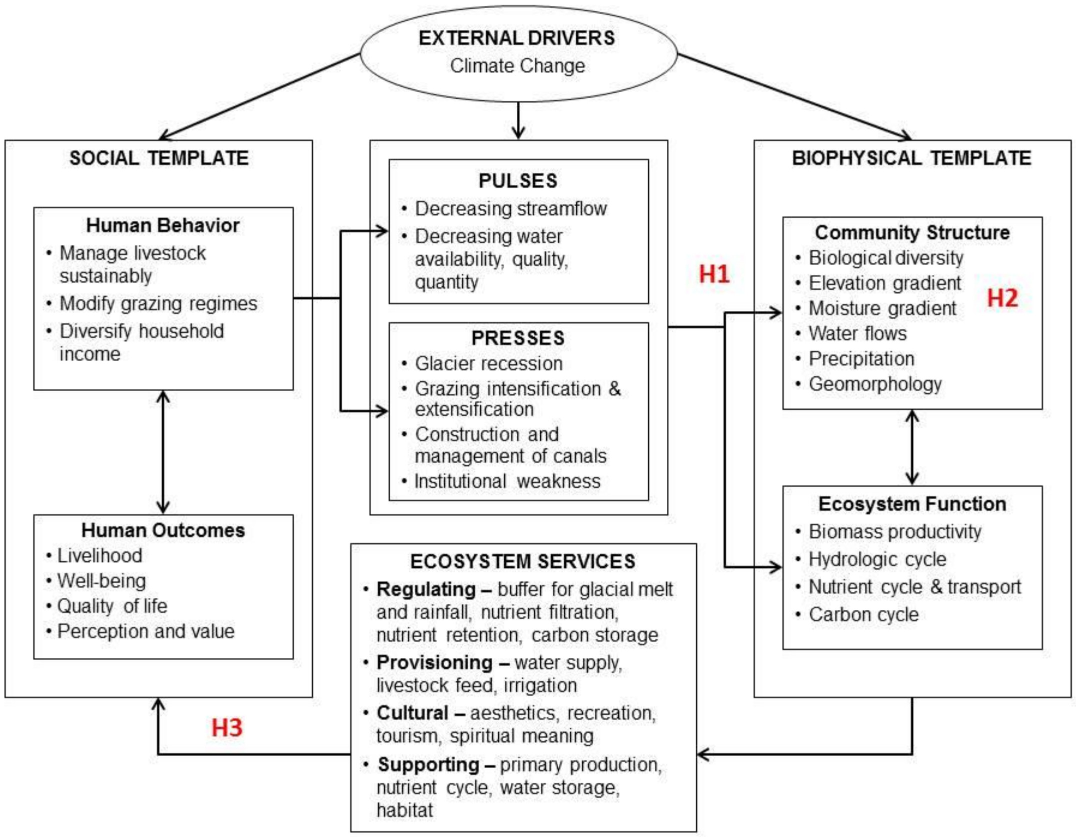


Figure 1.3. PPD framework adapted for Huascarán National Park with the three hypotheses for this dissertation identified in red.

In the PPD model, the biophysical template is shaped by disturbances, or presses and pulses (Smith, Knapp, and Collins 2009; Collins et al. 2011). The spatial configuration of peatlands in HNP is changing and likely affected by climate-induced glacier recession because meltwater influnces these ecosystems, a hypothesis I test and present in Chapters 3 and 4. The biophysical template represents processes that are described by principles from geology, hydrology, ecology, and biology. The biophysical template is divided into two components, community structure and ecosystem function. In the case of HNP peatlands, community structure includes biological diversity, elevation and moisture gradients, geomorphic processes, and precipitation; all these factors could explain patterns of heterogeneous vegetation composition that is evaluated in Chapter 5, where I address Hypothesis 2. The ecosystem functions of high Andean peatlands include biomass production, nutrient cycling and transport, carbon dynamics, and the hydrologic cycle.

Over time, presses and pulses would shape the biophysical template that then affects the quality and quantity of benefits that ecosystem services provide to humans (Collins et al. 2011). There are numerous ecosystem services that have been identified with high Andean peatlands (including *páramos*): pasture for grazing, carbon storage, biodiversity conservation, water supply for urban centers, agriculture and hydroelectric power generation, temporary water storage, groundwater recharge, sediment accretion, nutrient filtration, and pollution removal (Charman 2002; Buytaert, Cuesta-Camacho, and Tobón 2011). The Park's Master Plan lists additional ecosystem services: microclimate regulation, water purification, and migratory bird habitat (SERNANP 2011). Furthermore, high Andean wetlands have aesthetic and recreation value because the pastoral landscapes are visually appealing to tourists and local users. Peatlands and adjacent flat or gently rolling areas inside the park often serve as camping sites and can be heavily damaged by overuse. Based on field observations, camping likely has a negative impact on peatlands through heavy concentrated use and problems with waste disposal. In addition, peatlands are susceptible to disturbances resulting from overuse, grazing, and trampling (Pérez 1993). Non-native livestock (cattle, horses, sheep) are the major disturbance factor that could negatively impact the biophysical template and affiliated ecosystem services because the activities of grazing and trampling inhibit plant growth and reproduction and contribute to soil erosion.

In the PPD model, disturbances can be influenced by external drivers such as climate change, as is the case in HNP and evidenced by melting glacier ice and shifting hydrology. Environmental change is increasing the vulnerability of local communities who are adapting to new conditions (Young and Lipton 2006; Mark et al. 2010). The social template in the PPD model is comprised of human behavior and human outcomes. Human behavior refers to decision-making and actions, whereas human outcomes correspond to quality of life, perceptions, values, well-being, and livelihood (Collins et al. 2011). As defined by Chapin, Folke and Kofinas (2009), livelihood is defined as a strategy undertaken by individuals or social groups to create or maintain a living. They define well-being as an individual or social group’s quality of life, which is dependent upon meeting basic material needs for a good life. Local people’s livelihoods in the high valleys of the Cordillera Blanca primarily rely on agriculture and livestock. Other household economic activities may include tourism, manufacturing dairy products, and fabricating artisanal products, representing an income diversification strategy related to climate change (Mark et al. 2010). The PPD social template also includes human behaviors; managing livestock, modifying grazing regimes, and diversifying household income would all be considered human behaviors potentially important in the study area.

Livestock management decisions made by pastoralists impact the biophysical template through the grazing and trampling activity of livestock. What drives decisions and behaviors are values and perceptions of change, which I further address with empirical information in Chapter 6. Human outcomes of pastoralists in the Cordillera Blanca are simultaneously influenced by external drivers (climate change) and supported by ecosystem services. Livelihoods depend on livestock, which in turn depend on ecosystem services for survival.

The PPD framework is a relatively new and untested model that offers insight into how environmental changes are both impacted by and impact the social and biophysical systems in HNP. Central to geographic thought is the human-environment interaction paradigm and accompanying interdisciplinary engagement (Baerwald 2010). Because the PPD model integrates social and biophysical sytems and their intersections, I have designed an interdisciplinary approach to address my three research questions. Taken together, the three questions are larger than any one discipline, thus necessitating a research design that bridges distinct traditions to answer the research questions. Drawing from land change science, remote sensing, landsape ecology, spatial statistics, geographic information science, plant ecology, soil science, wetland science, multivariate ecological analysis, and ecological oral histories, the dissertation adopts multiple perspectives to produce new knowledge about a rapidly changing landscape related to glacier recession and therefore contributes to and advances geography.

DISSERTATION TIMELINE

The field data presented in this dissertation were collected over 5 seasons beginning in 2011 and culminating in 2015. All trips took place in July for approximately 30 days, with the exception of an 8-day trip in October 2015. A summary of the major accomplishments for each field season is presented below.

2011. In July, I conducted an initial trip to the study area with the objective of familiarizing myself with the area. It included a synoptic trip of the Santa River basin from the Pacific Coast to the headwaters of the basin at Conococha lake, spanning a 4000 meter elevation gradient. During this reconnaissance trip, I identified the three valleys that would likely be the focus of the research, Quilcayhuanca, Carhuascancha, and Llanganuco. I also began to identify the knowledge gaps, conceptualize the research questions and potential methods, determine feasibility of implementing the methods, and calculating the project budget.

2012. The primary objective of the second field season was to complete vegetation surveys in Quilcayhuanca and Carhuascancha and evaluate the feasibility of 50 cm soil pits (determined to be infeasible). I presented my proposed research at an international workshop at the Universidad Nacional Santiago Antúnez de Mayolo in Huaraz (in Spanish). I also conducted preliminary interviews with park staff and pastoralists.

2013. The third field season included two international conferences, the Foro Internacional de Glaciares, where I presented a poster. At the Glacial Flooding & Disaster Risk Management Knowledge Exchange & Field Training Workshop (organized by USAID and The Mountain Institute), I led a field-based module on mountain mires. I finished vegetation surveys in the Llanganuco valley and collected all the soil samples. In the Quilcayhuanca valley, I collected half of the soil samples.

2014. The fourth season was dedicated to collecting the remaining soil samples in Quilcayhuanca and all in Carhuascancha. I conducted ecological oral histories and hired and trained a field assistant and translator to conduct the oral histories after my field season ended. During this field season, I had the opportunity to develop a new collaboration with peat ecologists from Michigan Technical University. We cored peatlands in Quilcayhuanca, reaching up to 10 meter depths. The collaboration is ongoing and I expect to publish findings jointly in 2016. I also presented an update on my research to a public audience at the Museo de Historia Natural in Lima (in Spanish).

2015. The objective of this final trip was outreach where I shared the results of my dissertation with in-country partners and collected feedback. I presented to an audience of 30 people at The Mountain Institute in Huaraz and to 10 staff members of HNP (both in Spanish). The park staff is re-writing the Master Plan and, at their request, I submitted a summary of my finding that will be included in the 2016-2020 Master Plan. In addition, I presented to a graduate seminar on climate change and coupled natural human systems at the Universidad Pontifica Universidad Catolica del Peru in Lima (in Spanish). The conclusions in Chapter 6 reflect feedback received during these sessions and ideas generated during interactive discussions.

AUTHORSHIP

The research and writing presented herein is solely the product of the author, Mary H. Polk (*nom de plume* Molly H. Polk). Where collaborators contributed to data collection, their names are clearly indicated. In the future, individual chapters will be re-crafted into manuscripts to be submitted for publication in peer-reviewed journals where chapters will appear as multi-authored manuscripts. Following traditions in the discipline of Geography, I will be the lead author to signify that I conceptualized the research design, collected the data, performed the analysis, and drafted most, if not all, of the manuscript. This “effort-based” style lists subsequent authors in order of their relative contribution to the manuscript (Winkler 2014).

DISSERTATION OVERVIEW

Highlighted in red text in Figure 1.3 are the three hypotheses associated with the specific research questions that guide the dissertation. Chapter 2 provides an in-depth description of the study area, addressing both social and biophysical details at the scale of the valley and the park. Rather than describe the study area in each of the chapters, the reader is encouraged to refer to Chapter 2 for each subsequent chapter. Chapter 3 investigates the changing spatio-temporal character of peatlands in the park, with special attention to patterns and processes of change and ecological consequences of change. After documenting how peatlands are changing spatially over time, Chapter 4 then explores why peatlands are changing using an analytical technique adopted from econometrics. Next, Chapter 5 evaluates the community structure of peatland vegetation, concentrating on plant diversity and identification of the abiotic factors that explain vegetation heterogeneity. Finally, Chapter 6 delves into the important social aspects of peatlands through the use of ecological oral histories as a technique for estimating landscape change. Chapters 3, 4, and 5 provide results that explicitly treat biophysical systems in the PPD model. In contrast, Chapter 6 offers a social dimension that enriches the narrative by bringing forward local knowledge as a way to integrate perceptions of change into the research. It also concludes with recommendations for management strategies that would maintain and protect peatlands into the future.

Chapter 2: Study Area

This chapter describes the area of interest that is the subject of this dissertation, Huascarán National Park (8.5° – 10° S) and its glaciated mountain range called the Cordillera Blanca. The study area is defined by two scales: the park level and the valley level. At the park level, I include the entire 3400 km^2 contained within the legal park boundaries. Within the park, there is a finer-grain focus on three valleys, Llanganuco, Quilcayhuanca, and Carhuascancha (Figure 2.1, 2.2). Having two scales facilitates valley-to-valley (inter-valley) and within-valley (intra-valley) analysis as well as evaluating park-wide trends. Furthermore, when inter- and intra-valley level questions are addressed, they are easily contextualized within the broader scale of the park.

The selection of Llanganuco, Quilcayhuanca, and Carhuascancha offers a comparative approach. The 3 valleys span east-west and north-south gradients and they represent a variation of glacier coverage summarized in Table 2.1. The research design is based on a space-for-time swap selection strategy where each valley represents a phase in time along the continuum of glacier recession. The strategy is a proxy for sequential changes in time and enables valley-to-valley comparisons (Mark et al. 2010; Bury et al. 2011). Landscapes are the product of biophysical and social forces operating at global, regional, and local scales over time (Young et al. 2007) and I use this concept as an organizing framework for the study area chapter. It begins with an overview of relevant biophysical processes and then turns to social factors pertinent to the study area.

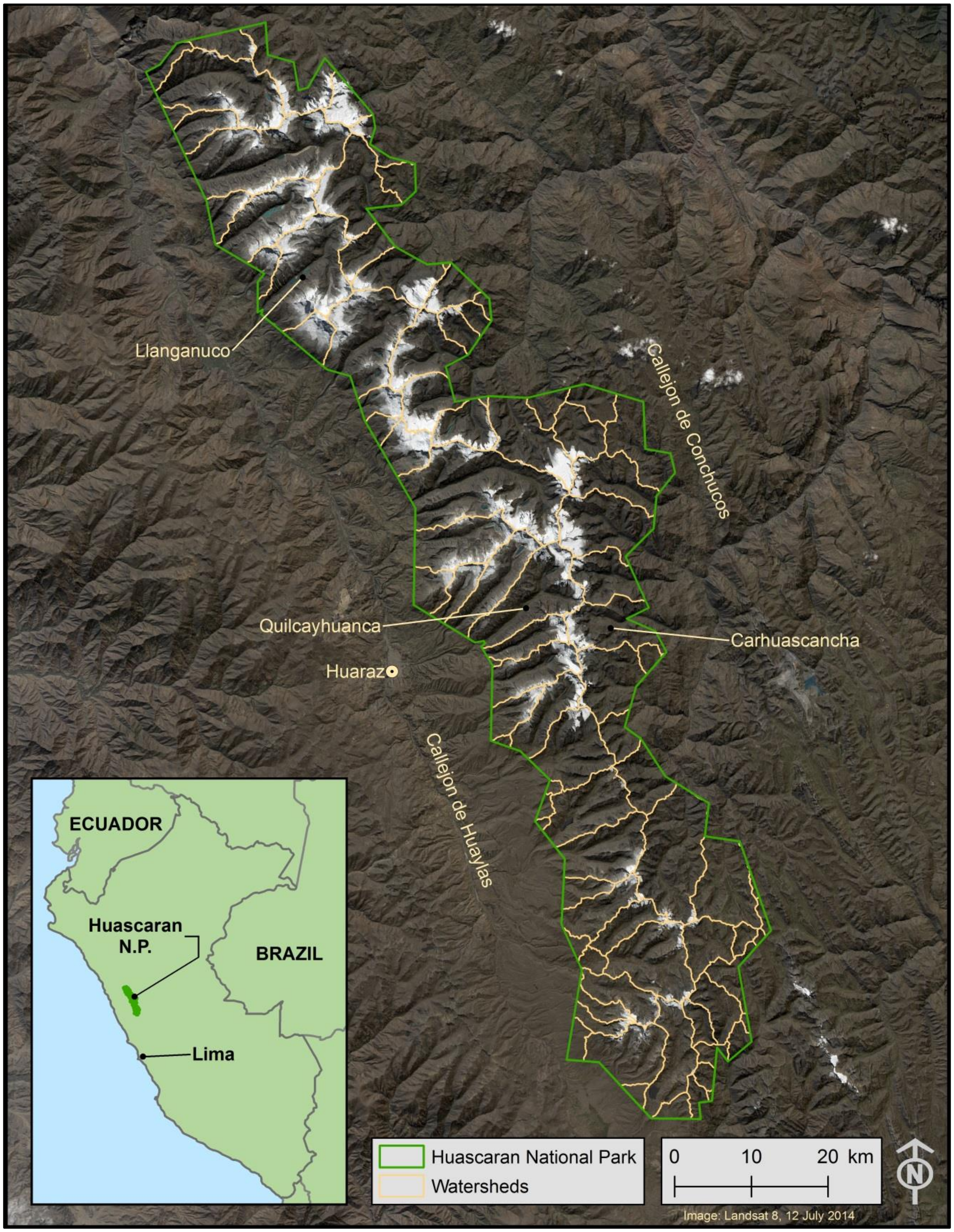


Figure 2.1. Study Area showing Huascaran National Park, glaciated Cordillera Blanca mountain range, and 3 focus valleys, Llanganuco, Quilcayhuanca, and Carhuascancha.

Table 2.1. Characteristics of the three study area valleys. Llanganuco and Quilcayhuanca information from (Mark et al. 2010) and Carhuascancha from author's GIS data.

	Llanganuco	**Quilcayhuanca**	**Carhuascancha**
Valley Area (km^2)	85	243	96
% Glaciated	36%	17%	8%
Side of Cordillera Blanca	West	West	East
Drainage	Santa River to Pacific Ocean	Santa River to Pacific Ocean	Mosna River to Amazon River and Atlantic Ocean
Latitude Longitude	9° 02' S 77° 36' W	9° 29' S 77° 24' W	9° 28' S 77° 14' W

Figure 2.2. Images of the three valleys, Llanganuco (Top), Quilcayhuanca (Center), and Carhuascancha (Bottom)

Biophysical Template

Geology

The Cordillera Blanca is a conspicuous feature of the Andes because it is home to the world's largest concentration of tropical glaciers (Kaser and Osmaston 2002). It is located in a section of the Andes that is comprised of three parallel mountain ranges with the Cordillera Blanca's ridgeline as the continental divide. The eastern slopes of the Cordillera Blanca drain into tributaries of the Marañon River, headwaters of the Amazon basin, and into the Atlantic Ocean. The western slopes drain into the Santa River that flows into the Pacific Ocean. The arid Cordillera Negra lies west of the Cordillera Blanca.

The Andes are formed by the collision of the oceanic Nazca plate with the continental South America plate, resulting in orogenesis. The denser Nazca plate subducts under the South America plate along the Peru-Chile trench at a rate estimated to be 8 cm/year (Kennan 2000; Lutgens and Tarbuck 2012). Initiation dates of subduction are debated, but one estimate places Andean orogenesis as early as the Triassic period (Coney and Evenchick 1994; Gregory-Wodzicki 2000; Canavan et al. 2014). Oceanic-continental collision and associated orogenesis is ongoing as evidenced by frequent seismic activity. Uplift rates vary over time and across the mountains' tremendous 5000 km length from Colombia to southern Argentina (Gregory-Wodzicki 2000).

The Western Cordillera of Central Peru, where the Cordillera Blanca is located, is described by Gonzales and Pfiffner (2012) as a belt comprised by Jurassic and Cretaceous sediments that folded and thrusted in the Paleocene to Eocene times. Batholiths, or exposed plutons, are a distinguishing geologic feature of the Cordillera Blanca (Sevink 2009) and can occur in subduction regions. As the descending plate melts, plutons are created and over geologic time, erosion and uplifting exposes the plutons. Cordillera Blanca batholiths intruded into the Jurassic and Early Cretaceous sediments during the Miocene, between 8 and 10 million years ago, making the mountain range fairly young in geologic chronologies (Gregory-Wodzicki 2000; Gonzalez and Pfiffner 2012). Approximately 80-90% of the Cordillera Blanca batholith is granodiorite, with the remainder attributed to isolated pods of tonalite and diorite. The batholith intruded into Jurassic metapelites from the Chicama formation (McNulty et al. 1998). The Chicama formation consists of dark grey and black shales that are commonly pyritic siltstone, quartzite and volcanics in some area (Cobbing et al. 1981). Sulfide-rich lithologies are present in the Cordillera Blanca and glacier retreat has newly exposed substrates to weathering. Sulfide and pyrite oxidation of the exposed substrates is linked to pH values below 4 in one glacier-fed stream flowing into the Santa River (Burns, Mark, and McKenzie 2011; Fortner et al. 2011).

Amidst the ongoing geologic processes, the east and west sides of the Cordillera Blanca are incised by glacial and fluvial activity. The range is transected by valleys that are perpendicular to the range's northwest-southeast orogenic strike. Glaciers occur at the heads of most valleys (known as *quebradas* locally), the origin of rivers that drain down these valleys. Because the range has the largest concentration of tropical glaciers in the world, glacial erosion rates are higher than nearby non-glaciated areas (Garver et al. 2005). Glacial scouring and fluvial erosion play a role in the geomorphology of the mountain range and in the formation of high altitude wetlands that line the valley floors.

Soils

Typically, mountain soils are described as young, poorly developed, thin and rocky (Price and Harden 2013), but they can also include deep ancient organic soils from peatlands (Hall et al. 2009). In the Cordillera Blanca, soils fall into one of three orders: entisols, inceptisols and histosols. Entisols are thin, poorly developed, and found on steep slopes and exposed ridges. They are shallow, usually well-drained, and show little evidence of horizons. Some entisols can be accumulations of fine or slightly rocky material (Price and Harden 2013). They are young and therefore weakly developed soils where the soil-forming processes began relatively recently (Brady and Weil 2002). Inceptisols are soils that are in the early stages of development; they are a family that includes soils with diagnostic horizons that fail to meet the criteria for other orders (Price and Harden 2013). Histosols are a third order of soils found in mountains. They are peat soils that form where drainage is poor, usually in depressions where water can persist. Histosols are characterized by an abundance of dead plant material, high acidity, and high organic matter content. The mineral content – albeit low – is derived from the erosion of upland slopes. They do not contain permafrost within a meter of the surface (Price and Harden 2013). Rodbell (1993) reported that peat soils exposed in a lake sediment section in a moraine in the Cordillera Blanca were dated to ca. 13,300 ± 200 yr. B.P. Table 2.2 reports the soil orders, suborders and total area found within Huascarán National Park according to the park master plan (SERNANP 2011). Although the park master plan does not recognize the presence of histosols, puna ecoregions do contain them (Young et al. 1997) and soil analyses made during this study indicate the presence of highly organic soils that would likely be classified as histosols (see Appendix 5 for soil analysis results).

Table 2.2: Soil orders, suborders and total area in HNP reported by SERNANP 2011. Note that SERNANP does not include histosols that are present inside the park.

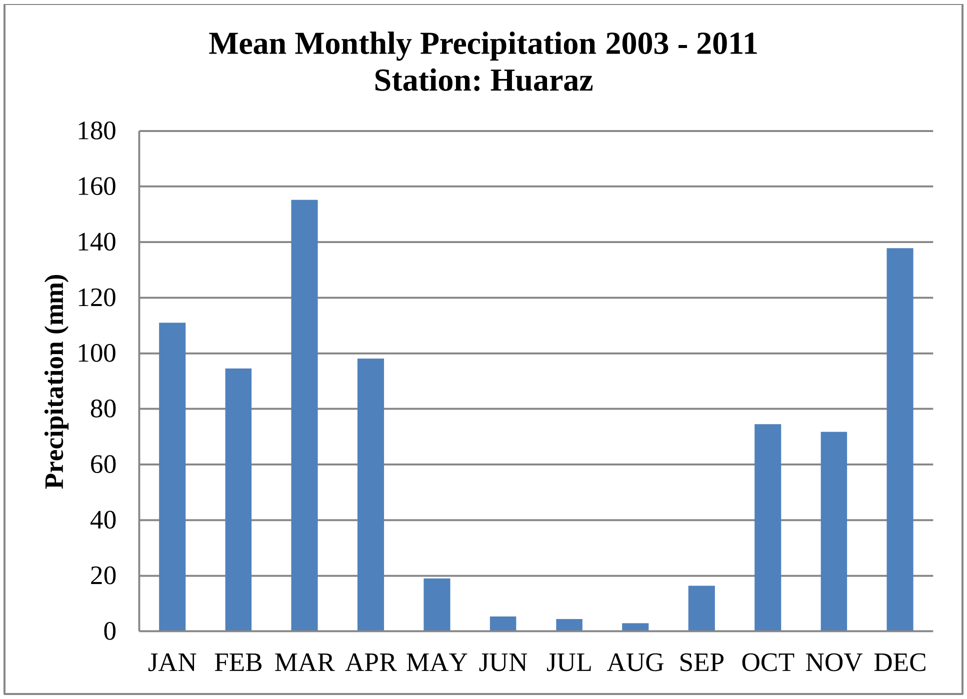
Soil Order	Soil Suborder	Area (km^2)
-	Rock Outcroppings	4.5
Entisol	Rock Outcroppings and Lithic Ustorthents	4.6
Entisol	Rock Outcroppings and Typic Cryorthents	1.7
Entisol	Typic Ustifluvents	1.3
Entisol	Typic Cryofluvents	2.5
Entisol	Typic Cryorthents	1.2
Entisol	Typic Cryorthents and Typic Cryofluvents	1.0
Entisol	Typic Ustorthents and Rock Outcroppings	1.2
Entisol	Typic Ustorthents and Typic Ustifluvents	1.8
Entisol and Inceptisol	Typic Cryorthents and Typic Cryumbrepts	2.8
Entisol and Inceptisol	Typic Ustorthents and Lithic Haplustepts	4.5
Entisol and Inceptisol	Typic Ustorthents and Typic Haplustepts	4.5
Entisol	Typic Endoaquents	1.5

Climate and Weather

Two climatic features drive the climate and weather in the study area, the orographic effect and the latitudinal movement of the Inter-Tropical Convergence Zone (ITCZ). The Cordillera Blanca is a high mountain barrier located perpendicular to the zonal air flow from the Amazon. The mountains block most of the easterly air flows that carry moisture from evaporation over the Atlantic Ocean and Amazon basin. When air masses meet the Cordillera Blanca, they are forced aloft where cooler temperatures trigger precipitation that falls on the mountains (Espinoza et al. 2009). West of the Cordillera Blanca, the neighboring Cordillera Negra and coastal plan receive little to no rainfall (Kaser et al. 2003) whereas precipitation levels on the eastern or windward side of the Cordillera Blanca can be 2 - 3 times higher than the leeward side (Johnson 1976). It should be noted, however, that there is strong spatial variability in precipitation patterns in the inter-Andean valleys east of the Cordillera Blanca particularly in valleys lower than 2000 masl (Espinoza et al. 2009).

The climate is characterized by distinct dry and rainy seasons. The annual pattern corresponds with the north-south migration of the ITCZ. The convergence of easterly trade winds results in intense convective activity that produces precipitation over the tropics. Easterly trade winds deliver moisture originating in the Amazon basin to the Cordillera Blanca (Garreaud, Vuille, and Clement 2003; Kaser et al. 2003). Thus the wet season is a period of high precipitation, cloud cover, and humidity (Schauwecker et al. 2014). Approximately 90% of precipitation occurs between October and April with a peak in February and March; the dry season is typically from May to September (Vuille, Kaser, and Juen 2008).

Unfortunately, there is no complete meteorological dataset for the 1960 – 2012 period for the Cordillera Blanca (Lipton 2008; Vuille, Kaser, and Juen 2008; Mark et al. 2010; Schauwecker et al. 2014). Nevertheless, researchers have reported that there is a north-south precipitation gradient; more rainfall occurs in the northern section of the study area than in the south. Precipitation at a northern meteorological station in the Parón valley is 770 mm/year versus 470 mm/year at the southern Recreta station (Vuille, Kaser, and Juen 2008). Precipitation also varies with elevation and records show that low elevation meteorological stations such as Caraz (2286 masl) receive substantially less precipitation than high elevation stations such as Parón (4185 masl) (Schauwecker et al. 2014). As is typical of tropical mountain temperature patterns, there is a wide daily variation and small annual temperature variation (Kaser, Ames, and Zamora 1990). Figures 2.3, 2.4, and 2.5 illustrate mean monthly precipitation and mean air temperature for the meteorological stations closest to the 3 valleys of interest as reported by Peru's national meteorological and hydrological service, Servicio Nacional de Meteorología e Hidrología del Perú (SENAMHI).



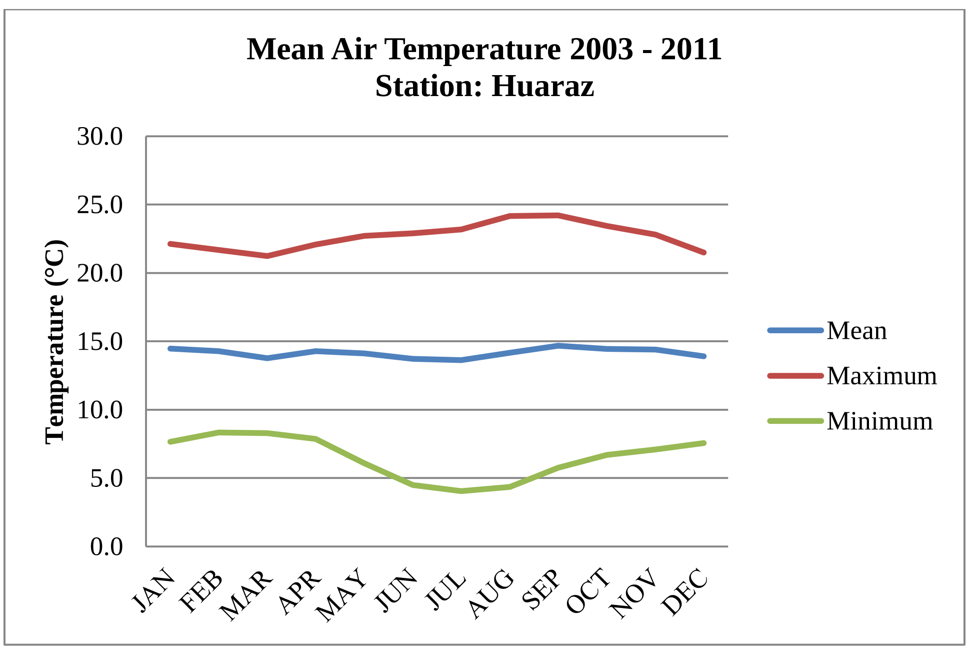
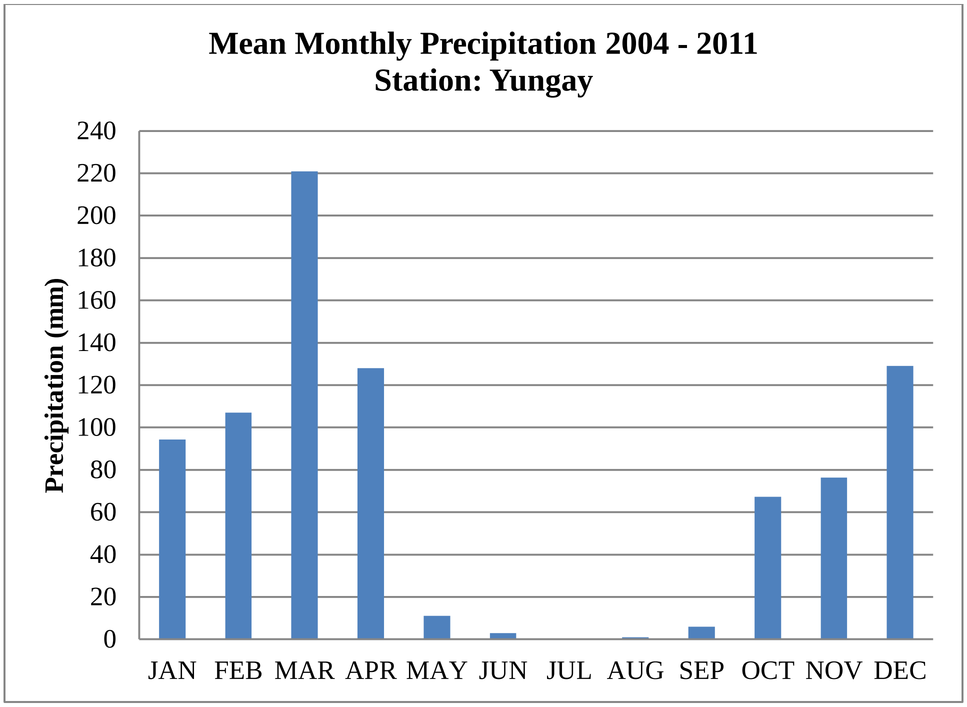


Figure 2.3. SENAMHI Precipitation and temperature data for Huaraz (2003 – 2011), corresponding with the Quilcayhuanca valley. Prior to 2003, data contains gaps.



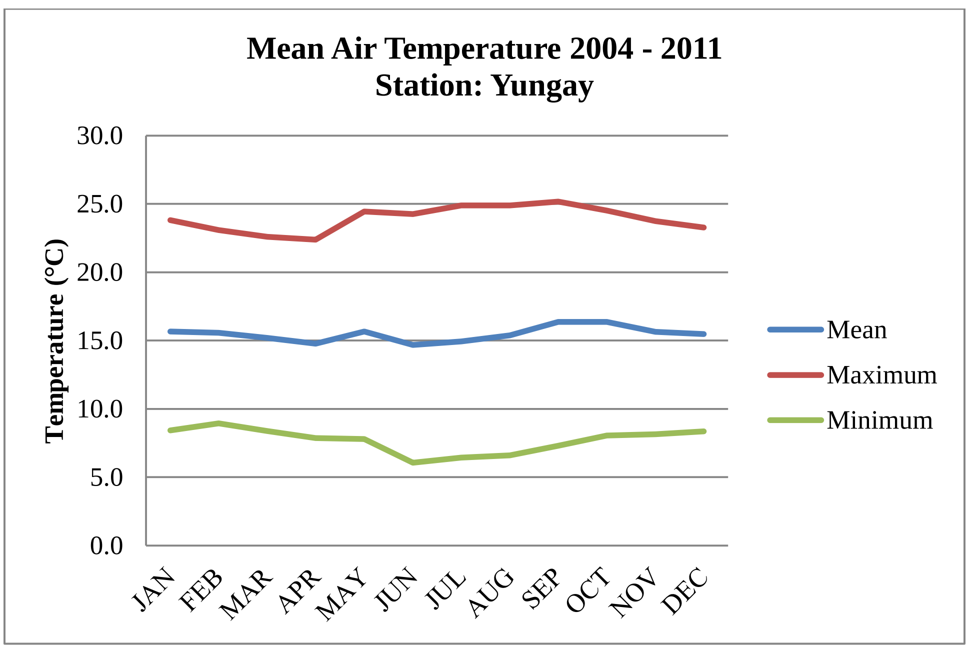
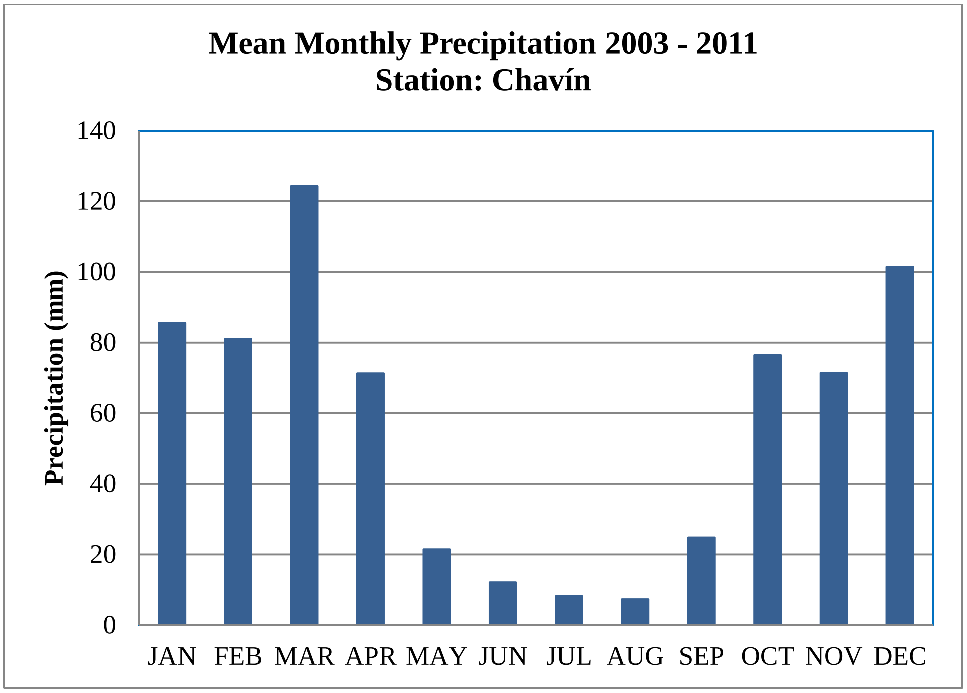


Figure 2.4. SENAMHI precipitation and temperature data for Yungay, 2004 – 2011, corresponding with the Llanganuco valley. Prior to 2004, data contains gaps.



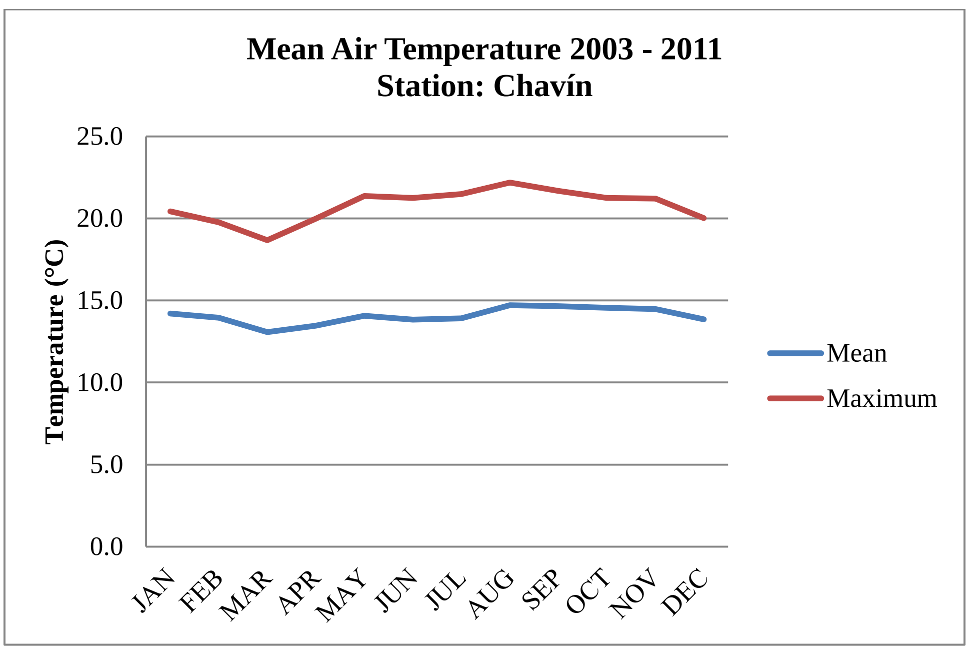


Figure 2.5. SENAMHI precipitation and temperature data for Chavín, 2003 – 2011, corresponding with the Carhuascancha valley. Prior to 2003, data contains gaps. Mean minimum temperature unavailable due to instrument failure (Vera 2014).

An additional climatological factor in the study area is the El Niño Southern Oscillation, or ENSO phenomenon. ENSO refers to the entire range of climatic variability including El Niño and La Niña events (NOAA 2014a). It is a coupled atmospheric and oceanic phenomenon that occurs roughly every 2 to 8 years (Garreaud et al. 2009). At the time of this writing, El Niño conditions are present. Authorities are currently predicting a 95% chance that El Niño will continue through the Northern Hemisphere winter 2015-2016 period (NOAA 2014b). Under normal conditions, southerly winds blow up the coast of South America and then turn west at the equator in a counterclockwise direction. These winds blow warm equatorial water south and west towards Australia and New Zealand. Cold water wells up from the depths of the Pacific Ocean just off the coast of South America in the Humboldt Current. The upwelling of cold water produces a nutrient rich habitat for marine life and an economically valuable fishery along the Peruvian, Ecuadorian, and Chilean coasts (Ruddiman 2008). The cold upwelling in the eastern Pacific and the pooling of warm water in the western Pacific is linked to the thermocline, a region of strong vertical temperature gradients in the uppermost section of the ocean (Caviedes 2007). Under normal atmospheric conditions, the Walker circulation drives east-west atmospheric circulation. In the western Pacific, air rises to create convective thunderstorms. Upper level winds then blow east, descending in the east Pacific and blows west as westerly trade winds (Collins et al. 2010). The Walker circulation is associated with a high pressure system in the eastern Pacific and a low pressure system in the western Pacific (Ruddiman 2008).

During an El Niño event, trade winds weaken and warm water that normally is in the western Pacific pools off the coast of South America, east of its normal position. As the winds weaken, the Humboldt Current is disrupted. The thermocline flattens and warmer water gathers where colder water normally occurs off the coast of Ecuador, Peru, and northern Chile. Sea surface temperatures in the eastern tropical Pacific increase by 2 – 5° C and sometimes as much as 8° C (Christopherson 2003; Ruddiman 2008). The Walker Circulation is altered and a low pressure develops in the east and a high pressure system exists in the eastern tropical Pacific (thus Southern Oscillation). A corresponding phenomenon to El Niño is La Niña, which occurs when surface waters in the equatorial Pacific off coastal South America are colder than usual (Christopherson 2003). The causes of El Niño and La Niña have not yet been identified and accurately predicting the onset remains elusive (Collins et al. 2010).

The effects of El Niño are not uniform across the Andes and vary considerably even within Peru (Rabatel et al. 2013). In the Cordillera Blanca, El Niño years correspond with above-average temperatures and decreases in precipitation. Consequently, during El Niño years, glaciers have a negative mass balance, whereas La Niña is associated with increases in glacier mass balance (Vuille, Kaser, and Juen 2008). El Niño events appear to correspond with less cloud cover thus exposing glaciers to higher solar radiation than normal years and enhancing negative mass balance (Smith, Mark, and Rodbell 2008). The teleconnection between Cordillera Blanca glaciers and El Niño, however, is considered “spatially unstable” and negative mass balance does not always occur during El Niño events (Vuille, Kaser, and Juen 2008, 14).

Fluctuating Tropical Glaciers

Tropical glaciers are found in New Guinea (Irian Jaya), the mountains of East Africa (Mt. Kenya, Mt. Kilimanjaro, Rwenzori) and in the Andes (Colombia, Ecuador, Bolivia, Peru). Of these, 71% are in Peru. Tropical glaciers are particularly sensitive to climate change. The absence of thermal seasonality in the tropical latitudes causes these glaciers to exhibit behaviors that are unlike their northern latitude cousins (Kaser 1999; Kaser and Georges 1999). Sensitivity to climate change can be partially explained by steep elevation gradients. Ablation occurs on the lowest part of the glacier year-round with higher ablation rates during the wet season; the glacier terminus is continually losing mass throughout the year. In the higher accumulation zone, snow is deposited only during the wet season (Kaser and Georges 1999; Rabatel et al. 2013). Tropical glacier snowlines are near the 0° isotherm, so any shift in temperature and precipitation can cause rapid mass balance changes. For example, a small increase in air temperature during the accumulation season means that precipitation that would normally fall as snow would fall as rain thereby decreasing mass balance (Rodbell, Smith, and Mark 2009). Kaser (1999) reported that a complex combination of changes in air temperature, humidity, precipitation, cloudiness and incoming shortwave radiation govern tropical glacier fluctuation. Yet, there is some disagreement: ice core evidence indicates that rising air temperature is the primary driver of tropical and subtropical glacier retreat and reduced precipitation levels are less important (Thompson et al. 2006). Research in the Cordillera Blanca by Mark and Seltzer (2005) attributes glacier recession to rising temperatures and accompanying increases in humidity. They conclude that changes in solar radiation (moderated by cloud cover) do not explain glacier recession, at least from 1962 to the present.

In spite of the ongoing discussion over what governs fluctuation, it is understood that tropical glaciers respond quickly to small shifts and to larger changes such as ENSO cycles (Rabatel et al. 2013; Schauwecker et al. 2014) and that Andean glacier fluxes are spatially heterogeneous (Rodbell, Smith, and Mark 2009). The Cordillera Blanca chronology of Holocene glaciation stands out as one of the best studied in the tropical Andes (2009). Evidence from the Pleistocence on the other hand is sparser. Rodbell (1993) refined work completed by Clapperton (1972, 1983) that dated Pleistocene moraines. Using rock-weathering posts on groups of moraines and modelling to generate a Pleistocene and early Holocene glaciation timeline, Rodbell (1993) estimates ages as far back as 590,000 yr B.P. A newer study provided evidence that there was extensive ice advance around 400,000 yr B.P. (Farber et al. 2005). Modelling places moraine dates ranging from 20,500 - 46,500 yr. B.P., 29,000 - 72,000 yr B.P., and potentially as old as 75,500 - 500,000 yr. B.P. (Rodbell 1993). Pleistocene advances are thought to be synchronous with regional and global glacial advances. Major advances at 29,000 yr. B.P. and 16,500 yr. B.P. reflect a cooling of tropical temperatures by > 4° C around the Cordillera Blanca (Farber et al. 2005).

Ice cores collected from the col of Huascarán by Lonnie Thompson and his team provides records from the Lateglacial Stage (LGS) through the Younger-Dryas transition (12,800 to 11,500 years ago) into the Holocene (Thompson et al. 1995) . During the LGS, tropical Atlantic sea surface temperatures may have been 5°- 6° C cooler and air temperatures in the high Andes were estimated to be 8°- 12° C cooler. In the Younger-Dryas, a global cooling event, glacier advance may have culminated between ~11,280 and 10,990 yr B.P. (Rodbell and Seltzer 2000) or earlier between 14,500 to 12,900 yr B.P. (Jomelli et al. 2014). Stansell and others (2013) recreated Holocene climate conditions using lacustrine sedimentary layers from lakes in Peru's Western Cordillera. They conclude that the early Holocene (~12.0 – 8.0 ka) was a period when glaciers retreated because of less precipitation. In the middle Holocene (~8.0 – 4.0 ka) the climate became colder and wetter, resulting in advancing glaciers. The late Holocene (after ~4.0 ka) was a period when glaciers retreated in response to increased temperatures and precipitation. During the Medieval Climate Anomaly (~1.0 – 0.7 ka), drier and warmer temperatures prevailed and glaciers once again retreated. Sediment records from elsewhere in Peru are in agreement – the MCA was a time of prolonged aridity (Bird et al. 2011).

All tropical glaciers (Irian Jaya, Mt. Kenya, Mt. Kilimanjaro, Rwenzori, and the Andes from Venezuela to Bolivia) were at maximum extent during the LIA and since the end of the LIA, they have all retreated. LIA fluctuations of tropical glaciers are thought to be globally synchronous by Kaser (1999). The LIA was a time of cooler, wetter climate conditions and tropical glaciers advanced (Bird et al. 2011; Rabatel et al. 2013; Stansell et al. 2013). Two major glacial advances in the Cordillera Blanca occurred during the LIA: one around AD 1330 and a second around AD 1630, which is believed to be the maximum LIA advance (Jomelli et al. 2009). Other smaller LIA advances occurred between AD 1590 and 1720, and between 1780 and 1880 (Solomina et al. 2007).

From the LIA maximum glacial extent to the beginning of the 20th century, Cordillera Blanca glacier tongues retreated approximately 1000 m or roughly 30% of their length, most likely responding to anthropogenic climate change (Vuille et al. 2008). The Huascarán ice cores show a distinct and rapid warming trend over the last 200 years (Thompson et al. 1995). After about 1880, significant and accelerated recession began (Mark and Seltzer 2005; Vuille et al. 2008). Since 1900, aridity in Peruvian Andes has been more pronounced than any other time during the preceding 2300 years (Bird et al. 2011). Equilibrium line altitudes have risen by about 70 m since 1969. Focused research on climate change in the latter half of the 20th century suggests that glacier recession is driven by increased temperatures and humidity levels. Reduced cloud cover and enhanced solar radiation are not thought to be forcing mechanisms (Vuille et al. 2003; Mark and Seltzer 2005). Temperature and humidity levels have increased throughout the tropical Andes from 1950 to 1998 and are linked to increases in tropical sea surface temperatures in the Pacific Ocean that raise tropical freezing heights (Vuille et al. 2003).

An analysis by Schauwecker and others (2014) of recent (1969-2012) temperature data from meteorological stations around the Cordillera Blanca shows that temperature increases are slowing. The running 30-year mean temperature decreased from a maximum of 0.31° C per decade between 1969 to 1998 to 0.13° C per decade from 1983-2012. Simultaneously, minimum temperatures increased and maximum temperatures cooled effectively decreasing the daily temperature range. Meanwhile, mean annual precipitation has strongly increased by roughly 60 mm/decade since 1980. The increase in precipitation would have led to glacial advance yet the strong increase in temperatures appears to have outpaced the precipitation increases; these dynamics appear to have fueled glacier retreat from 1969 to 1998. Glaciers may still be responding to pre-1998 increases in air temperatures (Schauwecker et al. 2014). Consequently, the outlook for Cordillera Blanca glaciers is pessimistic because air temperatures and Pacific sea surface temperatures are expected to continue increasing (Vuille et al. 2008; Jomelli et al. 2011; IPCC 2013).

Peruvian government findings are consistent with independent scientific studies documenting substantial decreases in glacier area (Kaser and Osmaston 2002; Mark and Seltzer 2005; Silverio and Jaquet 2005; Racoviteanu et al. 2008; Burns and Nolin 2014). The government completed their first detailed countrywide glacier inventory in 1970 and in the Cordillera Blanca total glacier area was 723.37 km^2. By 2003, total glacier area decreased to 527.62 km^2, a loss of 195.75 km^2, or 27% over 33 years. The remaining glaciers have not only receded, but also have fragmented. Small glaciers have in some cases disappeared altogether (i.e. the Broggi glacier). Glacier fragmentation has increased the number of glaciers: in 1970 there were 722 and there were 755 in 2003 (Autoridad Nacional del Agua, Unidad de Glaciologia 2013).

Changing hydrologic regimes and corresponding shifts in water resource use are some of the consequences of glacier recession in the Cordillera Blanca and throughout the Andes (Bradley et al. 2006; Chevallier et al. 2011). Historical hydrologic data used to model predictions about glacier discharge and streamflow in the Cordillera Blanca have produced inconsistent findings (Baraer et al. 2012). For example, Juen et al. (2007) and Vuille et al. (2008) posit that as glacier volume decreases, the volume of glacier discharge will also decrease. In another example, Pouyaud et al. (2005) suggest that streamflow increases as the glacier recedes and then, after reaching a critical threshold, decreases suddenly. Baraer et al. (2012) modified and refined previous concepts to suggest a new model. After analyzing discharge from nine watersheds in the Cordillera Blanca, Baraer et al. (2012) found that hydrological responses to deglaciation are nonlinear. Discharge increases, peaks and then declines gradually over time in a 4 phase conceptual hydrographic sequence. Furthermore, individual watersheds in the Cordillera Blanca are presently in different phases of deglaciation and associated phases of decreasing discharge. Seven of the 9 watersheds have passed peak discharge and have entered a declining phase (Baraer et al. 2012).

The Puna Ecoregion and Vegetation Formations

An inclusive definition of puna incorporates all habitats and vegetation types above 3300 masl and from 7° to 18° S. It is a tropical mountain system that is above the elevational limit of continuous forest but below the permanent snow line (Young et al. 1997). Carl Troll (1968) recognized three types of puna - wet, dry, and moist - based on differences in precipitation gradients from north to south and from east to west in the Andes. The majority of HNP is in the puna ecoregion and would be considered moist puna, which is mostly found on Peru’s western cordilleras (Young et al. 1997; SERNANP 2011). The Cordillera Blanca and HNP are at the transition of the puna to the south and the *jalca* ecosystem to the south, making it biologically diverse and a compelling place to work on vegetation and floristics (Smith 1988).

The floristic boundary between *jalca* and puna is ambiguous and some consider *jalca* to be a transition zone between páramo and puna (Luteyn 1999; Josse et al. 2011). High altitude mountain grasslands in northern Peru that receive high annual precipitation are termed *jalca* (Weberbauer 1945; Molina and Little 1981). Puna and *jalca* are similar to páramo, an ecosystem found in Ecuador, Colombia, and Venezuela that receives higher annual precipitation and is characterized by giant rosette plants, tussock grasses, acaulescent rosette plants, cushion plants, and sclerophyllous shrubs (Buytaert et al. 2006; Lüttge 2008).

The puna ecoregion has a long history of human influence from early use by hunter-gatherers (Baied and Wheeler 1993). Across the puna, humans have transformed the composition of large herbivore herds from wild deer and camelids to domestic camelids (llamas and alpacas), sheep, and cattle (Molina and Little 1981). Introduction of cattle, sheep, horses, pigs, and donkeys by humans predates the establishment of the national park in 1975. Populations of wild deer and vicuña are small. Montane grasslands in the Andes are heavily influenced and modified by anthropogenic activity including burning, and overgrazing (Young et al. 2007).

Taxonomic work in HNP is scattered and ranges from a PhD dissertation (Smith 1988) to studies on unique high altitude plant adaptations (Cano et al. 2010) to field guides. A recent master's thesis documented over 100 wetland and aquatic species growing in and around Conococha, a lake located immediately outside the southernmost boundary of HNP (Ramirez Huaroto 2011). Cataloguing the abundant and diverse tropical mountain flora continues and new species are still being discovered (Al-Shehbaz, Cano, and Trinidad 2012). In total, there are about 901 plant species and 374 genera in the park represented by 114 families including the iconic bromeliad, *Puya raymondii* (SERNANP 2011). Plants are adapted to low relative humidity and cold night-time temperatures. They often have adaptations to discourage grazing, such as prickles, spines, and thorns; the ability to re-sprout is a survival strategy against grazing and fire (Young et al. 1997). In terms of fauna, the park is home to 210 bird species, 25 mammals, 2 amphibians, and 4 reptiles, all adapted to high mountain environments (SERNANP 2011).

Several vegetation formations typify the puna ecoregion and some of these formations are found in HNP. Following is a description of each puna vegetation formation collated from varios sources (Molina and Little 1981; Smith 1988; Young et al. 1997, 2007; Cano et al. 2006, 2010; Lipton 2008; Josse et al. 2011; SERNANP 2011).

- Woodlands. These open montane forests are dominated by trees and large shrubs from the genera *Polylepis*, *Buddleja*, *Gynoxis*, and *Baccharis*. Most of the largest woodland patches are located on the eastern side of the park. Included in the woodland category is the non-native *Eucalyptus*, found at lower elevations along the park boundary closer to human settlements. Woodlands are often 10 – 20 m in height.
- Shrublands. Known locally as *matorrales*, these formations are low-growing evergreen woody shrubs that can encroach onto neighboring grasslands in the absence of fire. These plants have multiple woody stems and heights are generally 1-2 m and < 3 m. They tend to be found on rocky, well-drained soils and can be

on slopes or flat terrain. Common species include *Cassia hookeriana, Tecoma sambucifolia, Escallonia resinosa,* and *Baccharis* spp.

- Grasslands. The sweeping expanses of tussock or bunch grasses (called *ichu* by local residents) are perhaps most characteristic of puna. These open spaces comprise most of the puna and are dominated by graminoids (grass-like) plants such as sedges and grasses. Abundant genera from the Poaceae family include *Calamagrostis*, *Agrostis*, *Festuca*, *Stipa* and *Poa*. Admixed among the dense, compact tussocks are forbs. *Puya raimondii* dot the puna grasslands in small isolated patches.
- Aquatic and Semi-aquatic communities. Found in lakes, margins of lakes and rivers, areas of permanently standing water, these communities can be found at a variety of elevations. Important plant families are Asteraceae, Crassulaceae, Cyperaceae, Isoetaceae, Juncaceae, Poaceae, Potamogetonaceae, and Ranunculaceae. Aquatic plants are adapted to conditions where they are continually submerged and species richness is overall lower. Margins of lakes and rivers include communities of emergent aquatics, primarily Cyperaceae and Juncaceae.
- Wetlands. Called *bofedales*, *oconales*, or *humedales* locally, these water saturated systems may be peat forming or non-peat forming. In the case of the former, peatlands include both bogs and fens depending on the hydrologic source. In the case of the latter, these wet meadows have mineral soils. HNP peatlands are formed by dense cushion plants and rosettes, but *Sphagnum recurvum* is found

only in small patches in the study area. Peatland plant communities are low-growing (< 15 cm) and evergreen. Wet meadows are dominated by graminoids. Additional details are provided in the following section, Wetland and Peatland Ecosystems.

- Lithic Areas. These barren, rocky areas are mostly found above grasslands and below snow and ice fields. They also contain rock-falls and scree slopes. In lower elevation zones (3500 – 4000 m), mosses and ferns are abundant. Lichen such as *Rhizocarpon* may partially cover rocks and other lithic substrates. Lithic areas are well dispersed throughout the park and include moraines and zones that have previously been occupied by glaciers. Recently exposed substrates represent an opportunity for pioneering plants to initiate primary succession. Highly specialized plants (e.g. *Stangea henrici*) have elastic roots adapted to the daily movement of cryoturbed soil. Plants in these areas are often located in microsites that offer protection from the wind but allow for enough sunlight for photosynthesis.
- Snow and Ice Fields. These areas occur in the highest elevations of the park, from 4800 m to over 6000 m. They are covered in snow and ice. Included are debris-covered glaciers. Snow and Ice fields are free of vegetation except in rare instances where pioneering species might take root on debris-covered glaciers, but these are quite rare (I observed one in the Llaca valley) and have not been documented in the scientific literature.

Wetland and Peatland Ecosystems

Wetlands in HNP include peat forming and non-peat forming systems; both can be seasonally inundated and some peatlands are permanently wet. Examples of non-peat forming wetlands are those found in the Llanganuco valley. Soils are primarily mineral in contrast to the high organic matter typical of peat soils. Soil textural class veers towards clay loam and clay silt. Non-peat forming systems can be considered wet meadows, wetlands that are dominated by herbaceous plants rooted in occasionally flooded soils (Keddy 2010). Both the non-peat forming wetlands and peat-forming wetlands are used for grazing by local communities, but it is unknown if animals preferentially select peat or non-peat forming wetlands.

Wetlands in the study area can be peatlands, referring to areas that are covered by peat, a highly organic soil comprised of the remains of plant and animal matter that has not decomposed completely (Rydin and Jeglum 2013). Peatlands are considered to be a wetland type where the plant matter accumulation rate exceeds the decomposition rate. Wetlands are considered peatlands when at least 30 cm of peat has accumulated (Martini, Martínez Cortizas, and Chesworth 2006; Rydin and Jeglum 2013). Water levels must be relatively stable and peat soils are often permanently waterlogged (Keddy 2010). Other authors have noted that high altitude wetlands in the Andes are usually peatlands (Earle, Warner, and Aravena 2003; Clausen et al. 2006; Cooper et al. 2010).

The largest peatlands in HNP line the floors of U-shaped valleys (Figure 2.5a). They are dissected by streams and rivers that originate either from glaciers or springs. The peatlands are lawn or carpet-like in their appearance with occasional patches of hummocks that are 10-20 cm in diameter and 15-25 cm tall. To walk on, the peatlands are firm, but can be soft and quake when water saturation is high. On the valley floors they are flat to slightly sloping (≤ 5°). Smaller sloping peatlands, or soligenous peatlands, are found on valley sides where steep walls transition into flat valley floors. Springs and seeps create conditions where the constant presence of water encourages peat formation. Slopes up to 26° have been observed. Peatland disturbances include impacts from livestock activity – trampling, overgrazing, and excavation for geophagy – as well as human activities such as canal construction and maintenance, trails, and camping.

Both fens and bogs are present in HNP, forming a fen-bog complex. Fens are minerotrophic and minerogenous – the primary hydrologic input is groundwater. In contrast, bogs are ombotrophic and ombrogenous receiving hydrologic input from precipitation. Fens have a higher pH than bogs, usually > 4, and their soil nutrient content is higher. As a result, they are floristically richer than bogs. The water table is at the soil surface or slightly below and they have slow internal drainage. Peat depth is greater than 30 cm, but can be shallow at the margins of the peat body where it transitions to mineral soils. Bog soil depth is always greater than 30 cm and the soil surface is generally higher than the surrounding topography, thus isolating it from surrounding mineral soils through vertical accumulation of organic matter. Convex features, both large and small, can be found on bog surfaces. Bogs are more acidic than fens, with a pH ≤ 4. They are nutrient poor and have lower species richness than fens (Keddy 2010; Rydin and Jeglum 2013).

Initially, peatlands may be fens supported by groundwater moving directionally downslope. Over time, as the peat body grows vertically, parts of it become isolated from underlying geology and groundwater inflows (Rydin and Jeglum 2013). The fen can evolve towards a raised bog, a system that is hydrologically controlled less by groundwater and more by precipitation (Keddy 2010). Convex topography characterizes bogs and these features in HNP can be 1-3 m above the surrounding peatland. While raised bogs derive input from precipitation, the adjacent and connected fens are controlled by groundwater. Combinations of fens and bogs in close connection to one another may lack distinct patterning features and are referred to as mosaic mixed mires or fen-bog complexes (Rydin and Jeglum 2013), and typify those in HNP. For the purposes of this dissertation, peatlands in the study area will be referred to as fen-bog complexes (Figure 2.6).

Figure 2.6. Examples of HNP peatlands from the Llanganuco valley (L) and a fen-bog complex in the Quilcayhuanca valley (R).

Peatland hydrology is influenced by the distinct seasonality of the tropical climate. Thus both permanent and seasonal peatlands are found in the study area. During the wet season peatlands can be inundated by overland sheet flow from precipitation. Hummocks and hollows can contain standing water during the wet season, but during the dry season, soil may be moist. There is no overland sheetflow during the dry season and soil moisture is spatially heterogeneous, ranging from dry to fully saturated. Plants that compose the carpet-like peatlands in the study area are adapted to permanently wet soil conditions as well as seasonally-driven water fluctuations. Obligate species that characterize permanent wetlands are *Plantago tubulosa*, *Eleocharis albibracteata*, and *Oritrophium limnophyllum*. Facultative upland species that dominate seasonal peatlands include *Lachemilla pinnata*, *Agrostis breviculmis*, *Lucilia kunthiana*, and the introduced non-native *Trifolium amabile*. Although permanent and seasonal wetlands may differ in terms of seasonal hydrology and species composition, they may be spatially contiguous (Young et al. 2007).

SOCIAL TEMPLATE

Efforts to create a national park were initiated in 1960 by the then senator of the Department of Ancash. After 15 years of planning and negotiations, including early participation by Peace Corps volunteers Curry Slaymaker and Joel Albrecht, Huascarán National Park was finally established in 1975 by Decreto Supremo N^{o} 0622-75-AG. The objectives for the park's creation were to protect the largest glaciated tropical mountain range in the world and its high biodiversity, unique geological formations, glaciers, and scenic beauty. Additionally, it protects archaeological sites and aims to develop tourism activities that provide socio-economic benefits to local communities. The park is part of the natural, cultural, and scientific national patrimony (SERNANP 2011). Peru's highest peak, Huascarán at 6768 masl, is a special point of national pride. UNESCO declared the park a Biosphere Reserve in 1977 and a World Heritage Site in 1985. The park has a core area of 3400 km^2, and a buffer zone of 1702 km^2. This study focuses exclusively on the core area in which all the high altitude wetlands are located.

Prior to the establishment of the park, jurisdiction was divided among large farms (*haciendas*), private ownership, or mining concessions (Young and Lipton 2006). Local communities control land use throughout numerous valleys in the park much as they did prior to the park's establishment. Detailed information concerning the park's establishment can be found in Lipton (2008) and Lipton (2014). Although biodiversity conservation purposes are the objective of the park, local communities continue to maintain land use rights within the park and its buffer zone. There are 43 registered Peasant Communities (*Comunidades Campesinas*) that use areas in the park for agriculture and grazing; these areas are managed communally. Thus, large sections of the park are subject to triple-layered governance: by communities, the regional government of Ancash, and the national government through the Ministry of the Environment (*Ministerio del Ambiente*) and its National Service of Natural Areas Protected by the State (*Servicio Nacional de Areas Naturales Protegidas por el Estado* or SERNANP). Multiple layers of governance can has led to conflicts over natural resource allocation, land management, and rights-to-use (Lipton 2008; Mark et al. 2010; Carey, French, and O'Brien 2012; Lynch 2012). For example, puna in HNP is used by communities for cattle, sheep, horses, and other livestock and their management goals do not synchronize with the park's biodiversity and conservation goals. This discord is an ongoing source of tension between Peasant Communities and park officials and full treatment of the multi-faceted conflicts are beyond the scope of this dissertation. Further information can be found in Lipton (2008), Mark et al. (2010), Carey, French, and O'Brien (2012), and Lynch (2012).

There are roughly 83,000 inhabitants in the combined park and buffer zone. In the park itself, there are 50 human settlements and about 550 inhabitants. Of the 83,000 inhabitants, 60% are on the east side in the Callejón de Conchucos and 40% are on the west side in the Callejón de Huaylas (SERNANP 2011). The park is subject to heavy pressure from surrounding communities. The Department of Ancash has 1,063,459 inhabitants (529,708 males, 533,751 females) and of those, 682,954 are urban dwellers and 380,505 are rural (INEI 2007). The major urban clusters around the park are Huaraz, Caraz, Yungay, Carhuaz, Recuay, Catac, Chavín, San Marcos, and Huari. There are approximately 267,000 people in the western Callejón de Huaylas and roughly 100,000 in the eastern Callejón de Conchucos (Mark et al. 2010). Most of the commercial activity and population is centered in Huaraz, the 2nd largest regional city and departmental capitol of Ancash. The regional economy is dominated by mining, commercial services, tourism, and manufacturing. Agriculture is an important component of the departmental economy; the major crops are potatoes, sugar cane, yellow corn, alfalfa, *maíz choclo* (large kernel corn), asparagus, and rice. Tourism is an important economic driver in Ancash: there were 944,100 tourist visits in 2011 and of those, 3.5% were foreigners (Banco Central de Reserva del Peru 2014).

In contrast to departmental trends, people living in and around the park are mostly engaged in smallholder agricultural production. Most livelihoods are dependent upon water and natural resources for agriculture and livestock production (Mark et al. 2010).

Livelihoods are based on diversified production strategies that include subsistence agriculture, pastoralism, mining and industry, tourism, and wage labor. Households often engage in a variety of these strategies to diversify income sources, satisfy basic needs, and strengthen resiliency (Lipton 2008). As stream discharge becomes more variable, following glacier recession and households must rely on unpredictable water sources. Accordingly, households are adding new diversification strategies – migration and remittances – to offset economic losses and increased risk. Migration to urban centers such as Chimbote and Lima permits younger household members to remit money to family members in the highlands. As a result, demographics are shifting and there is now a higher concentration of elderly people and fewer working-age people (Wrathall et al. 2014).

Chapter 3: Tropical Mountain Wetlands Shifting Through Space and Time

INTRODUCTION

Home to the world's largest concentration of tropical glaciers, Peru's Cordillera Blanca mountain range is a research node for investigators seeking to evaluate not only glacier recession, but also the coupled social and biophysical changes that are occurring (Young and Lipton 2006; Mark 2008; Chevallier et al. 2011; Bury et al. 2013; Carey et al. 2014; Wrathall et al. 2014), although there are many important research topics that have not yet been addressed. Tropical mountain wetlands that are downslope from and spatially proximal to glaciers are an ecosystem that should be in theory responding to glacier recession. The term wetland is used inclusively here to refer to highly organic fens and bogs as well as to minerogenous wet meadows.

The goal of this chapter is to characterize the spatial distribution of wetland change within Huascarán National Park (HNP) for a 23-year period (1987 - 2010) using remotely sensed data. To accomplish this goal, the chapter will begin with a brief exploration of overall landscape transformation before detailing wetland change. Understanding the broader context of change on the landscape level inside the park boundary is necessary in order to understand finer scale changes occurring in wetlands. After the landscape context, the chapter then focuses on wetland spatio-temporal heterogeneity by exploring trajectories of change based on landscape ecology metrics, including rates of change, and gains and losses for five time steps between 1987 and 2010. The findings highlight the importance of prioritizing wetland conservation and minimizing disturbance in order to maintain ecosystem integrity and carbon stores.

Landscape ecology, a discipline that connects spatial patterns and ecological processes (Turner and Gardner 2015), provides the theoretical framework for this chapter by applying the patch-corridor-matrix paradigm to land cover classes in the study area. The unit of analysis is the individual wetland, which is considered a patch in the puna matrix. Vegetation inside the park is a mosaic of high Andean plant communities, but dominated by grasslands called puna. Embedded in the puna matrix, wetlands form patches along valley floors. Using thematic products (categorical data) derived from Landsat TM imagery, I quantify spatial heterogeneity over a 23-year period. This longitudinal panel approach facilitates a temporal trajectory analysis in which trends or profiles of change can be constructed from landscape metrics (Crews-Meyer 2002, 2006). From these change trajectories ecological processes can be inferred. How the spatial configuration and composition of patches (and landscapes) change over time allows landscape ecologists to evaluate ecological structure and function (Kupfer 2011).

Central to landscape ecology is the role of disturbance, a key driver of spatial heterogeneity over time (Turner and Gardner 2015). A disturbance is defined as a discrete event that affect ecosystems, communities or population structures by causing changes in resources, substrate availability, or the physical environment (Pickett and White 1985; Chapin, Folke, and Kofinas 2009; Turner and Gardner 2015). Disturbances create and respond to spatial pattern at multiple spatial and temporal scales (Turner and Gardner 2015) and can be considered as presses or pulses (Collins et al. 2011). Wetland disturbances may stem from anthropogenic or natural sources (or combinations of both) and often result in hydrologic interruptions that change ecosystem properties (Keddy 2010).

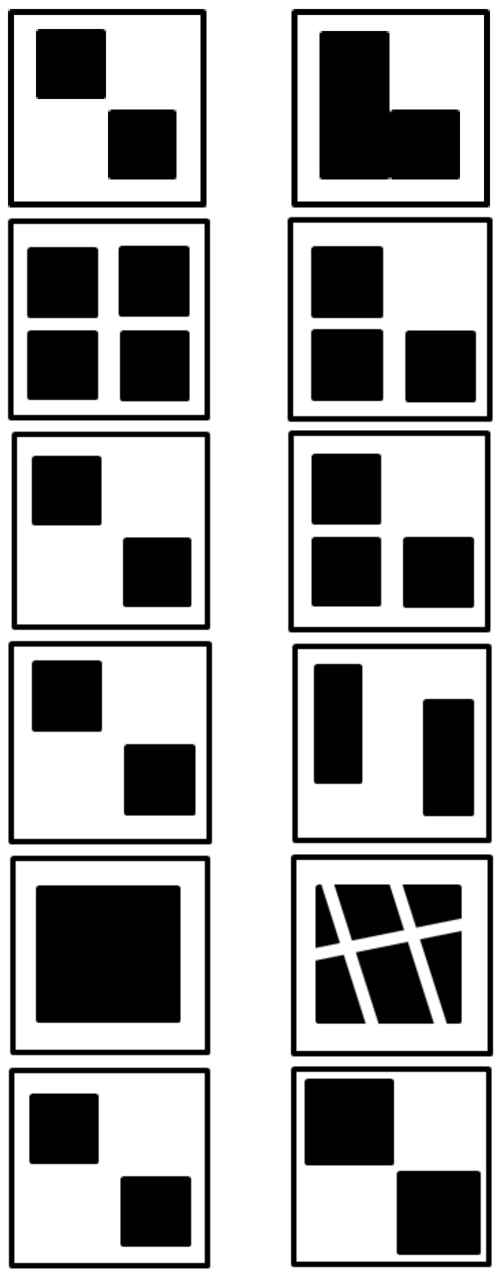
Regarding anthropogenic disturbance, Sluyter (1994) noted that geographers have led the way exploring the relationship between humans and wetlands, particularly in Latin American agricultural production where farmers drained wetlands to optimize soil moisture for maximum crop growth. In the Ecuadorian and Peruvian Andes, wetlands have been transformed for potato, tuber, maize and grain production, known as raised fields (Knapp and Denevan 1985; Turner II and Denevan 1985; Zimmerer 1991, 1994). Working in what is today southern Mexico and northern Central America, geographers have investigated upland and coastal wetland soils and water chemistry for evidence on ancient Maya adaptations to environmental change (Beach et al. 2009; Luzzadder-Beach and Beach 2009). Maldonado Fonken (2015) and Salvador and others (2014) identified human disturbance that negatively affect high Andean wetlands in Peru centered around resource extraction. Overgrazing, peat harvesting for cooking and heating fuel, mining, dam construction, and road construction form the dominant processes interrupting wetland functioning (Salvador, Monerris, and Rochefort 2014; Maldonado Fonkén 2015). One exception to this is in the south central Peruvian highlands, where human activities have actually helped create wetlands through canals that divert water to low-lying areas (Postigo, Young, and Crews 2008). Based on field observations during the summers of 2011-2014, the following anthropogenic activities were noted as disturbances directly affecting wetlands: overgrazing, trekking, camping, ditch construction and maintenance, and mining (limited to a few valleys). In other parts of Peru, protection provided by the government minimizes anthropogenic disturbances (Salvador, Monerris, and Rochefort 2014; Maldonado Fonkén 2015).

In terms of natural disturbances, this chapter focuses on the effect of long-term climate-driven disturbance – glacier recession – on wetlands inside the park. Climate change is expected to affect wetlands in significant ways (Keddy 2010). For example, changes in precipitation patterns, and more extreme climatic events will impact wetlands by changing hydrologic processes (Mitsch and Hernandez 2013). Climate change is predicted to trigger "variations in the structure, pattern, process, and function of wetlands by modifying temperature, hydrology, biogeochemical cycles, evapotranspiration, and shifting species distributions, altering community structures and species interactions" (Junk et al. 2012, 161). Because wetlands and climate are so closely linked via hydrologic processes, any changes in climate will alter not only wetland biodiversity, but also their extent and spatial distribution (Charman et al. 2008).

Within this context of anthropogenic and climate-related disturbances, I have identified four possible alternatives that could characterize spatio-temporal wetland change in HNP. The four alternatives combine concepts assembled by Bogaert and others (2004) from the landscape ecology literature. They identified ten common spatial processes and patterns in landscape transformation (Figure 3.1). The possible alternatives are as follows:

1) Wetland area could increase and patches could coalesce through the process of aggregation;

2) Wetland area may decrease and patches may undergo combinations of attrition (disappearance), deformation (change in patch shape, but not size), dissection (sub-dividing patches into smaller patches that are roughly equidistant), fragmentation (sub-dividing patches into unevenly separated patches), shrinkage (reduction in patch size, but without attrition) or perforation (formation of openings within patches);
3) Wetlands may experience stasis (no change in area) with patches persisting through time; and
4) Wetland area may change in a nonlinear fashion and exhibit combinations of land transformation processes. This alternative takes into account the possibility that combinations of all the land transformation processes may occur over the period of study. The spatial transformation processes may not be mutually exclusive between 1987 and 2010.



	Time 1	Time 2	Probable occurrence in HNP wetlands
Aggregation			Not likely
Attrition			Likely
Creation			Not likely
Deformation			Likely
Dissection			Likely
Enlargement			Not likely

Figure 3.1. Ten spatial processes in land transformation. Redrawn by the author from Bogaert et. al 2004.

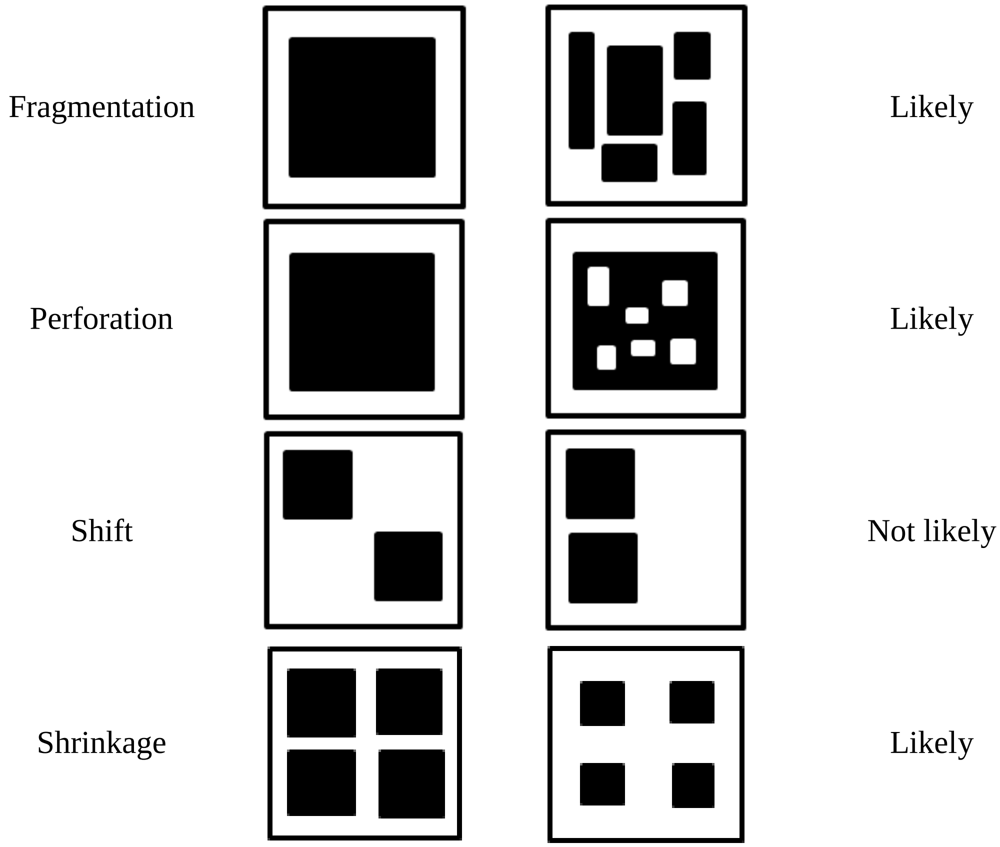


Figure 3.1. Continued. Ten spatial processes in land transformation. Redrawn by the author from Bogaert et. al 2004.

A review of the literature addressing spatial changes in high altitude wetlands shows that all four alternatives outlined above have been observed. In the case of expanding wetlands, Postigo and others (2008) reported that wetlands in central Peru expanded from 230 km^2 in 1990 to 983 km^2 in 2000, or 327%. Local livelihoods depend on sheep, llama, and alpaca herding; of these three livestock species, alpaca are adapted to grazing in wetlands. To benefit their alpaca herds, local communities build canals that flood soils that then expand wetland cover. In China's Altay Mountains, peatland area is increasing due to temperature and precipitation increases. Peatlands were 931.5 km^2 in 1990 and expanded to 977.7 km^2 in 2010, a 5% increase over 20 years. The authors theorize that melting glaciers could have contributed elevated hydrologic input to peatlands causing growth (Li, Xu, and Zhao 2014).

In an assessment of the vulnerability of tropical Andean ecosystems to climate change, Young and others (2011) predict that high altitude wetlands may shrink and disappear altogether following glacier recession. Bury and others (2013) showed that wetlands in the Quilcayhuanca valley in HNP decreased by 17% from 2000 to 2011 and became fragmented, presumably due to glacier recession and altered hydrology. Nie and Li (2011) measured change in high mountain wetlands near Mt. Everest in China. They found that wetlands measured 1678.8 km^2 in 1976, 1679.0 km^2 in 1988, and 1663.5 km^2 in 2006, equating to a 15.3 km^2 or ~1% decrease in total. Because some wetland increased, the total area of change (increase and decrease) added up to 94.5 km^2 or 5.6% between 1976 and 2006. Increases in temperature and decreases in precipitation are believed to be driving wetland change (Nie and Li 2011). In the Ruoergai Basin in northeastern Tibet, peatlands have been shrinking since the 1960s according to Li and others (2015). They speculate that ditch construction designed to drain peatlands accelerated erosion leading to losses in peatland connectivity (Li et al. 2015). In another study by Jiang and others (2014), wetlands along China's Heihe River exhibited significant fragmentation and perforation from 1975 to 2010; core wetland area decreased by 42.5%. The authors credit fragmentation and wetland loss to the combination of human activity and climate change (Jiang et al. 2014).

Peatland change in area may also be non-linear and can contain surprises as evidenced in paleoecological records. Peatland responses to climate change are not expected to be smooth and monotonic (Charman et al. 2008). Peatlands in the Zoige Basin on the eastern Tibetan plateau have shown patterns of decreasing and increasing area over a 20 year period. In 1990, peatland area was 4143 km^2, decreased to 3407 km^2 by 2000, and increased to 3589 km^2 in 2009. Changing temperatures and precipitation do not appear to be the cause; instead, a combination of factors explains nonlinearity. Growing populations increased demand for peat fuel mining and peatlands were drained to expand pasture for livestock grazing. In the late 1990s, protected areas were established and grazing areas were abandoned allowing peatlands to recover (Yao et al. 2011).

Ecosystems can exhibit persistence, or stasis through time that can be informative because they are locations of stability and resilience. Stasis (no change) may be caused by the absence of a driver of change and it may also be the result of two forces acting in opposition. Together, they may cancel each other out in terms of total area affected (Young 2008). In an example from wetlands on the Tibetan Plateau, Gao and others (2012) reported a net loss of 599 km^2 from 1994 to 2001. In 1994 there were 6780 km^2 and by 2001 wetlands had decreased to 6181 km^2. Some 4988 km^2 persisted and those were found on slopes from 2-9° (Gao, Li, and Brierley 2012).

Building on the studies reviewed above, this chapter uses similar methods including remote sensing, GIS, landscape ecology metrics, and rates of change to characterize spatio-temporal wetland heterogeneity. Whereas the seven studies above use satellite imagery from 2 or 3 acquisition periods (with the exception of Jiang and others (2014) who used 5 dates), this work uses 6 acquisition periods to ascertain a higher temporal resolution between 1987 to 2010. In and around the Tibetan Plateau, analysts using optical satellite imagery are faced with frequent and extensive cloud cover, a factor that has been a formidable challenge (Nie and Li 2011; Zhao et al. 2015). The effect of cloud cover has been reduced here by using only dry season imagery that is relatively cloud-free. An added benefit of exclusively using dry season acquisition dates is that phenological differences are minimized. An improvement to the previous studies is the application of ten distinct landscape transformation processes (Bogaert, Ceulemans, and Salvador-Van Eysenrode 2004) to characterize possible wetland spatio-temporal heterogeneity. Most authors focus on modifications to wetland area and only one author describes fragmentation (Jiang et al. 2014); however, wetland changes exhibit variations in spatial pattern that involve additional transformations that merit a more detailed framework such as the one published by Bogaert and others (2004). Finally, this chapter improves on the literature reviewed above by calculating net contributions to wetland change from other land cover classes using an approach published by Schulz and others (2010) to evaluate trajectories of forest change. Therefore, this chapter builds on previous work in high mountain wetlands by utilizing similar methods, but makes advances by incorporating techniques and approaches that have never before been applied to these unique ecosystems.

DATA AND METHODS

In this chapter, methods involved a two-stage process beginning first with a pixel-based image classification followed by calculation of the pattern metrics. A series of six Landsat TM images were downloaded from the USGS (http://glovis.usgs.gov) for 1987, 1990 1995, 1999, 2005, and 2010; each of the six time periods is comprised of two scenes (path-row 8-66, 8-67; Table 3.1). The earliest cloud-free image available is in 1987. Attempts were made to space images in roughly 5 year increments to capture landscape change on regular intervals, but in the case of 2000 cloud cover was extensive so it was necessary to use 1999. After June 2012, limited acquisitions were made by Landsat TM and the sensor was decommissioned in June 2013 (http://landsat.usgs.gov/L5_Decommission.php). In keeping with the attempted 5-year increment, the final image selected was 2010. To minimize the effects of intra-annual vegetation phenology all scenes used occur in the dry season (June, July, August). Although there are higher spatial resolution images available (i.e. SPOT, ASTER, GeoEye), the Landsat TM sensor has the longest series of earth observation images available, spanning ~30 years (http://landsat.usgs.gov/about_landsat5.php). Additionally, using only Landsat TM imagery allows the analyst to capitalize on sensor continuity and minimize cross-sensor and cross-platform effects (Jensen 2007). Images are Level 1T, a processing level that provides systematic, radiometric, and geometric accuracy by using a digital elevation model for topographic accuracy. The processing level is designed to optimize the ability to complete data intensive multi-temporal earth observation studies. Image-to-image registration is consistent across the dataset and is less than one-half of one pixel (Hansen and Loveland 2012). Image geometry for the 6 time periods was derived from the application of ground control points from the Global Land Survey 2000 (GLS2000) dataset. The GLS2000 is a library comprised of ground control points derived from elevation data collected by the Shuttle Radar Topography Mission and other elevation datasets (Gutman et al. 2008). Consistent geometry is essential in mountain areas because some topographic features are closer to the sensor than others, resulting in image distortion (Weiss and Walsh 2009). In the study area, Burns and Nolin (2014) previously showed that topographic correction did not systematically improve image classification when using an index-based threshold selection because available digital elevation models are error-prone; therefore additional topographic correction was not applied. Atmospheric correction was not performed because analysis was performed post-classification (Song et al. 2001; Jensen 2007).

Table 3.1. Landsat TM image acquisition dates. Each time period is comprised of two scenes, path-row 8-66 and 8-67, both collected on the same date.

Time Period	Acquisition Date
1	15 May 1987
2	10 July 1990
3	25 August 1995
4	20 August 1999
5	3 July 2005
6	18 August 2010

Image classification was performed in ERDAS Imagine 2014. Two scenes (path-row 8-66 and 8-67) for each date were mosaicked together. All 7 bands in 30 m spatial resolution were used in the hybrid supervised-unsupervised classification technique (Messina, Crews-Meyer, and Walsh 2000; Walsh et al. 2003), which has been used elsewhere in the Andes to measure land change (Kintz, Young, and Crews-Meyer 2006; Lipton 2008; Postigo, Young, and Crews 2008). The technique first employs an unsupervised classification method (ISODATA) to cluster the spectral information into

255 signatures. These signatures are then evaluated for separability using the transform divergence method, removing signatures with low spectral separability (<1950). Next, the edited signatures are applied in the supervised classification. The resulting files were filtered with a 4x4 pixel window to remove speckling. Adding ancillary data improves the accuracy of difficult-to-classify wetlands (Ozesmi and Bauer 2002; Adam, Mutanga, and Rugege 2009); a near infrared-red band ratio (Landsat TM bands 4 and 3) supplemented the classification. Lake color variation is wide in the images due to differing amounts of suspended glacial flour. These variations resulted in poor classification of the water bodies by the hybrid method. Attempts to improve lake classification using the Normalized Differential Water Index (NDWI) were also ineffective; therefore lake borders were manually digitized for each of the 6 time periods using the GLIMS dataset as a base layer (http://www.glims.org/). Known for its glacier inventory, the GLIMS dataset includes some lake features (Kargel et al. 2014). The thematic files for each date were finally coded into 7 land cover types: Barren, Puna, Wetland, Snow/Ice, Water, Shadow, and Cloud (refer to Table 3.2 for descriptions; there are no towns and very few built structures thus a “Built” class was omitted). These classes were modified from other land cover studies completed in the area (Lipton 2008; Silverio and Jaquet 2009), permitting comparisons. From this point onward, “Wetland” will be used to refer to the land cover class analyzed in this chapter, whereas “wetland” will be used to apply more broadly to the ecosystem.

Table 3.2. Seven land cover classes and descriptions.

Class	**Description**
Barren	Exposed geologic substrate void of vegetation
Puna	Tussock vegetation of bunch grasses and forbs admixed with patches of high Andean woodlands (i.e. *Polylepis, Gynoxis, Baccharis*) and woody shrubs.
Wetland	Zones where mesic conditions occur seasonally and permanently. Vegetation adapted to standing water or saturated soils. Includes peatlands (bogs and fens) and minerogenous wet meadows.
Snow/Ice	Areas covered by permanent snow and ice
Water	Primarily pro-glacial lakes, open water
Shadow	Shadow created by the sun angle and extreme topography typical in mountainous areas
Cloud	Atmospheric clouds or thick haze that obscure earth's surface

Accuracy assessments were performed on two of the six images: 1999 and 2010. Ideally, accuracy assessments would be performed on all images, but reference data that met accepted criteria (higher spatial resolution, acquired at approximately the same time of year, and with the same spectral resolution) were unavailable for all the time steps. A trial accuracy assessment for the 1987 image was performed using SPOT 1 imagery (4 scenes, acquired 28 May and 2 June 1987, 20 m spatial resolution, 2.3° incidence angle). The accuracy statistics for this trial were relatively poor (overall accuracy 65% and overall Kappa 60%) for two reasons. First, cloud cover and topographic shadow in the SPOT image differed from those in the Landsat image and introduced significant error in these two classes affecting the overall and kappa measurements. The combination of high topographic relief and the off-nadir viewing angle create extensive shadows in the SPOT 1 image. Second, SPOT 1 and Landsat TM spectral resolutions are dissimilar. SPOT 1 collects in three spectral bands (green 500-590 nm, red 610-680 nm, and near-infrared 780-890 nm) and offers a panchromatic band from 510-730 nm (10 m spatial resolution). In contrast, the Landsat TM sensor collects seven spectral bands (blue 450-520 nm, green 520-600 nm, red 630-690 nm, near-infrared 760-900 nm, near-infrared 1550-1750 nm, mid-infrared 2080-2350 nm, and thermal 10,400-12,500 nm at 120 m spatial resolution). Wetlands can be distinguished using the red and near-infrared bands (Ozesmi and Bauer 2002; Adam, Mutanga, and Rugege 2009) and the sensor band differences between SPOT and Landsat mean that wetlands appear differently, thus confusing the accuracy assessment. Consequently, the decision was made to use reference data exclusively from the Landsat ETM+ sensor, which share spectral similarities with Landsat TM. Landsat ETM+ offers a 15 m panchromatic band. In the case of 2010, the reference data were from the global Web-Enabled Landsat Data (WELD) project (http://globalweld.cr.usgs.gov/index.php), which produces cloud-free monthly, seasonal and annual composite images at 30 m spatial resolution. A WELD image from August 2010 was selected to correspond with the 2010 thematic image. Even though the WELD products do not have a spatially higher resolution, it was decided that some accuracy measurement was preferable to none. Reference data features are summarized in Table 3.3.

Table 3.3. Reference Data Summary.

	1999	2010
Sensor/Product	Landsat ETM+	WELD (Landsat TM and ETM+)
Acquisition Date	11 July 1999	August 2010 composite
Spectral Resolution	Panchromatic 520-900 nm	Band 1 520-600 nm Band 2 630-690 nm Band 3 780-860 nm
Spatial Resolution	15 m	30 m

Calculation of the Wetland pattern metrics for each of the 6 time steps was performed using FRAGSTATS version 4.2 (McGarigal et al. 2012) with the 8-neighbor rule specified, equating to a minimum patch size of 7200 m^2 or 0.72 ha. Because many pattern metrics are highly correlated (Turner, Gardner, and O'Neill 2001; McGarigal 2015), only several key metrics were selected (Table 3.4) based on prior use in wetlands and ability to characterize fragmentation, patch shape, and connectivity (Torbick et al. 2006; Zhang et al. 2009; Zhao et al. 2009; Kelly, Tuxen, and Stralberg 2011; Nie and Li 2011; Tovar, Seijmonsbergen, and Duivenvoorden 2013; Jiang et al. 2014; Liu et al. 2014). Annualized rates of change were computed for each time period and for the 1987-2010 period using a formula presented by Puyravaud (2003) and applied by Schulz et al. (2010) and Tovar et al. (2013) for purposes similar to this study. The annualized rate of change is derived from the Compound Interest Law and is calculated as follows:

$$r = (1 / (t_2 - t_1)) \times \ln(A_2 / A_1)$$

where r is the annual rate of change, t_2 is the second time period, t_1 is the first time period, A_2 is the area of the land cover class in t_2, and A_1 is the area of the land cover class in t_1. A GIS (ESRI ArcGIS 10.2 and Clark Labs Land Change Modeler) was used to evaluate the extent of changes among classes for each time step.

Table 3.4. Class metric descriptions for Wetlands adapted from (McGarigal 2015).

Metric (Acronym)	**Description (unit)**
Wetland Area	Amount of Wetland area (hectares).
Wetland Proportion (PLAND)	Sum of the area of all Wetland patches divided by total landscape area and multiplied by 100 (percent).
Number of Patches (NP)	Total number of patches in the landscape (no unit). NP is a simple measure of the extent of fragmentation.
Mean Patch Area (AREA)	Average Wetland patch size (hectares).
Total Edge (TE)	Sum of the lengths of all Wetland patches (meters). TE is an absolute measure of edge length. TE increases with increasing fragmentation.
Euclidean Nearest Neighbor (ENN)	Distance to the nearest neighboring Wetland patch based on shortest edge-to-edge distance measured from cell center to cell center (meters). Used to quantify patch isolation.
Shape Index (SHAPE)	Equals patch perimeter divided by the square root of patch area and adjusted by a constant to adjust for a square standard. Measures the complexity of patch shape compared to a standard shape (square) of the same size (no unit). Range is ≥ 1 without limit. SHAPE is a measure of shape complexity.
Cohesion Index (COHESION)	Equals 1 minus the sum of patch perimeter divided by the sum of patch perimeter times the square root of patch area for patches of the corresponding patch type, divided by 1 minus 1 over the square root of the total number of cells in the landscape and multiplied by 100 to convert to a percentage (no unit). Quantifies the connectivity of habitat as perceived by organisms dispersing in binary landscapes. Ranges from 0 – 100. Cohesion increases as the patch class becomes more physically connected.
Core Area (CORE_MN)	Mean patch core area. The interior area of Wetland patches after an edge buffer is eliminated. Two edge buffers of 10 m and 50 m were specified because the edge effects in peatlands are unknown.

RESULTS

Accuracy Assessment

Overall classification accuracy for the 1999 image was 91.5% and kappa was 89.8% (Table 3.5). Classification accuracy for the 2010 image was 78.49% overall and Kappa was 74.23%. Although the 2010 assessment values are less than the desired 80% benchmark, Producer's and User's accuracies were greater than 80%, with the exception of the Producer's value for Puna, which was 58%; Producer's and User's accuracies for Shadow were 70.59% and 41.86% (Table 3.5). Producer's accuracy is a measurement of omission, or the probability that a reference pixel is correctly classified. User's accuracy is a measurement of commission, or the probability that a classified pixel matches the same reference pixel. Differences in Shadow between the WELD product and the classified image lowered the overall and kappa values. Most of the confusion occurred between Shadow and Puna and Wetland (refer to Appendix 2 for confusion matrices).

Table 3.5. Accuracy Assessment results.

	1999		**2010**	
Class	**Producer's (%)**	**User's (%)**	**Producer's (%)**	**User's (%)**
Barren	89.36	84.00	81.82	79.75
Puna	83.78	93.00	58.00	80.56
Wetland	97.62	82.00	80.00	80.00
Snow/Ice	89.09	98.00	85.71	100.00
Water	98.99	98.00	98.67	96.10
Shadow	92.16	94.00	70.59	41.86
Overall (%)	91.50		78.49	
Kappa (%)	89.80		74.23	

Overall Landscape Transformations

In order to understand Wetland change, it is necessary to first explore the larger context within which it is occurring. The results of overall landscape transformation reveal not only that numerous changes are occurring to land cover classes individually, but also that there are interactions among these classes. Because the focus of this chapter is on Wetland change, a comprehensive in-depth analysis of landscape transformation is beyond the scope of the chapter. In this section, I present a brief summary of overall landscape change that provides an overview of land cover dynamics inside the park. The results of the overall landscape change are large and warrant dedicated treatment in a separate manuscript. Doing so would fully address the many changes and complex interactions among classes. All metrics for all classes are reported in Appendix 3. Following this section, I present Wetland results.

Table 3.6 and Figures 3.2, 3.3, and 3.4 report landscape composition through time. The results show that the HNP matrix is Puna, a class that includes Andean grasslands, shrublands and woodlands. Puna dominates the landscape composition and accounts for 151,418 – 193,867 hectares between 1987 and 2010, or 44.5 – 56.9% of the overall landscape. The second largest class is Barren, or exposed geologic substrate, occupying 53,309 – 95,172 hectares, or 15.6 – 28.0% of the landscape. Third is Snow/Ice, which ranges from 45,850 – 68,905 hectares, equivalent to 13.5 – 20.2% of the landscape. Wetland is the fourth largest land cover class ranging from 12,402 – 15,703 hectares, or 2.0 – 4.4% of the landscape. Water, representing pro-glacial lakes, ranges from 2421 – 2854 hectares, or 0.7 – 0.8% of the landscape.

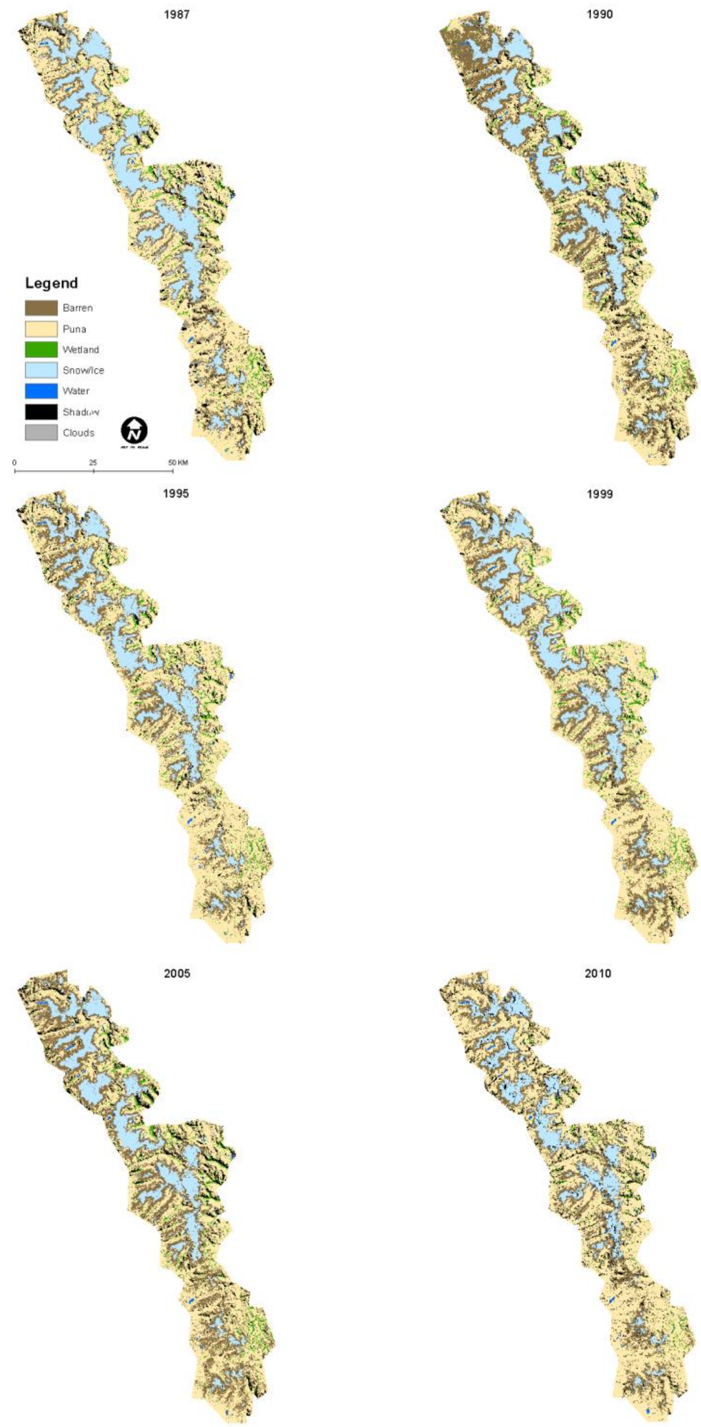


Figure 3.2. Classified images for all 6 time steps.

Table 3.6. Class areas over time (hectares).

	1987	1990	1995	1999	2005	2010
Puna	180,252	151,418	187,679	177,281	156,704	193,867
Snow/Ice	68,905	59,955	55,961	55,327	49,512	45,850
Barren	53,309	90,604	64,384	82,335	95,172	63,781
Wetland	12,402	13,985	13,858	15,703	14,166	6821
Water	2421	2392	2488	2649	2790	2855
Shadow	21,394	22,170	16,328	7899	21,286	27,486
Cloud	1849	0	0	0	918	0

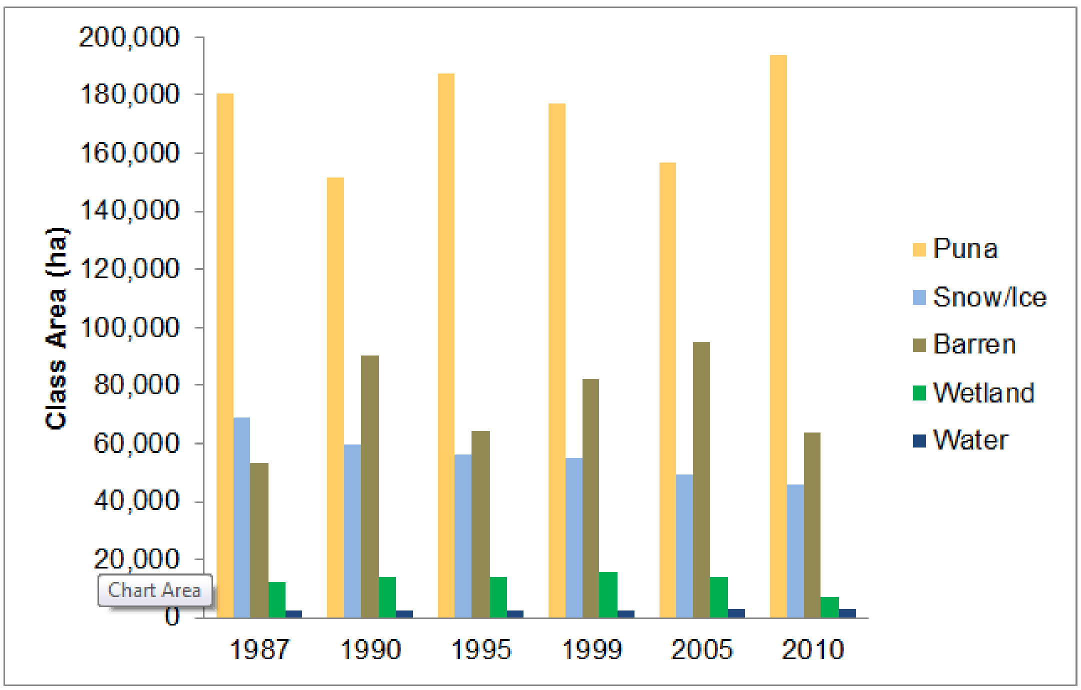


Figure 3.3. Class areas for the 6 time periods expressed in hectares. Shadow and Cloud are omitted.

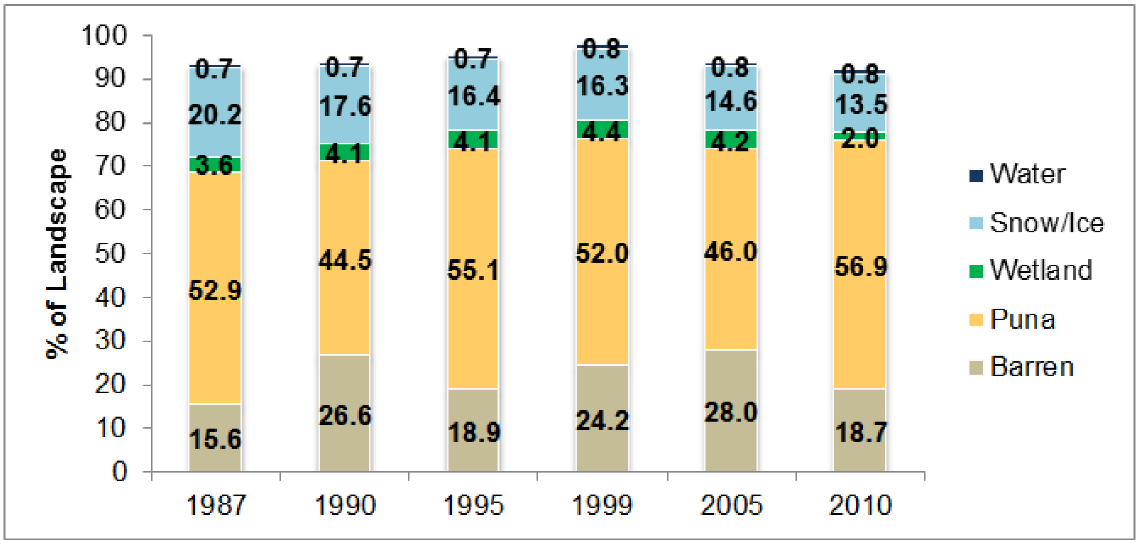


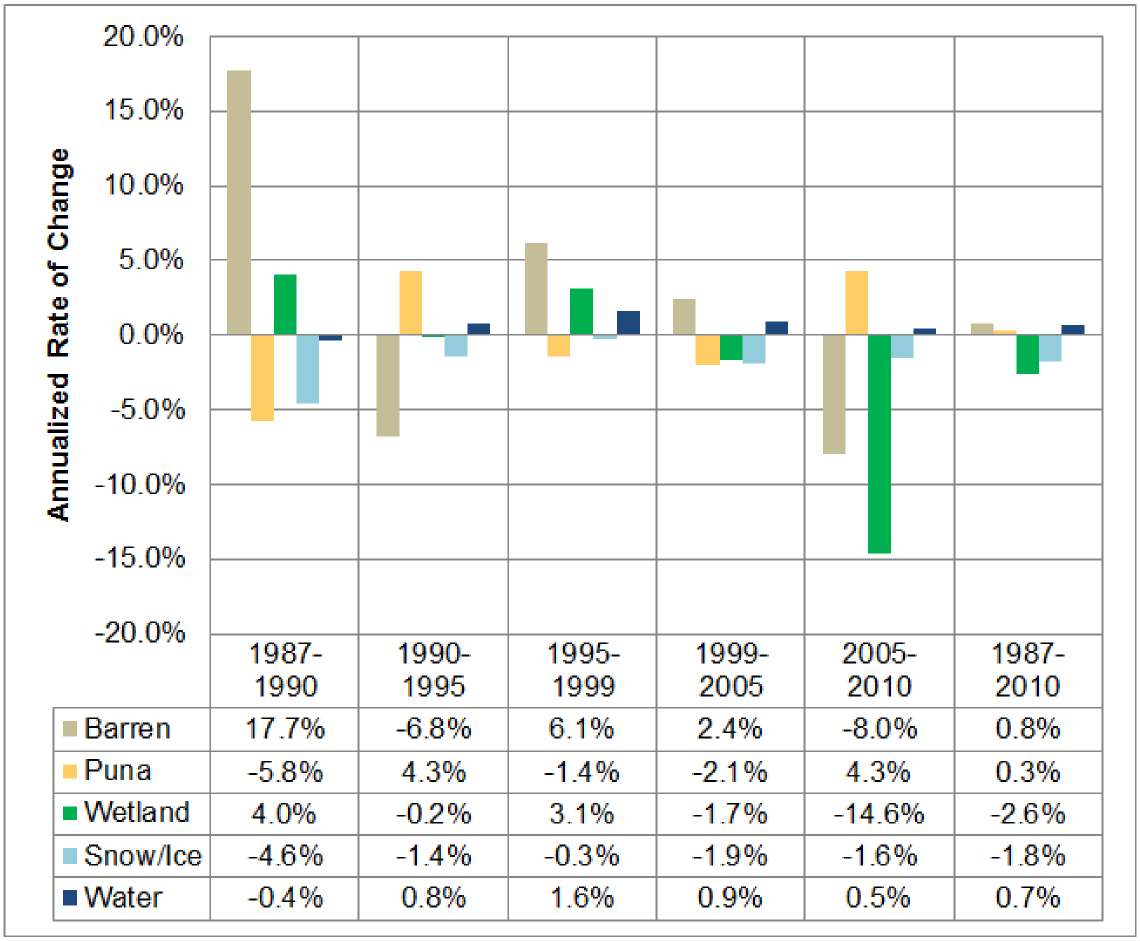
Figure 3.4. Landscape composition for the 6 time periods expressed as proportions. Totals are <100% because Shadow and Cloud are omitted.

Between 1987 and 2010, the land cover classes in HNP exhibit varying rates of change for each of the five time steps (Figure 3.5). The largest magnitudes of rates of change occur in the Barren and Puna classes, both experiencing increases and decreases in area throughout the 23-year period. Annualized rates of change also increase and decrease during each time step; Barren rates of change range from -8.0% to 17.7%, whereas Puna ranges from -5.8% to 4.3 (Figure 3.5). Changes in Barren and Puna are closely linked and are likely coupled to decreasing Snow/Ice cover (see Discussion). Figure 3.6 illustrates the relationship between Puna and Barren, with synchronous increases and decreases during each time step.

Snow/Ice decreased by 33% over the study period from 68,905 to 45,850 ha (Table 3.6). Rates of change are consistently negative and are -1.8% per annum, varying between -0.3 to -1.9% (Figure 3.5). These results are consistent with other published estimates for the same study area (Silverio and Jaquet 2005; Racoviteanu et al. 2008;

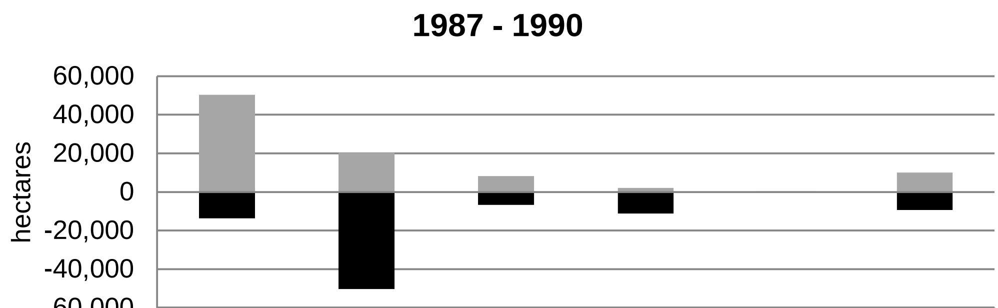
Baraer et al. 2012; Burns and Nolin 2014). Figure 3.6 illustrates that there is some gain in Snow/Ice during each time step, but losses outpace gains consistently. The gains are likely due to the ephemeral presence of seasonal snow on slopes adjacent to permanent ice fields. The remote sensing techniques used herein are unable to distinguish between snow cover and permanent ice because of their similar spectral characteristics.

Water, primarily representing proglacial lakes, occupies between 0.7 to 0.8% of the landscape. It exhibits a fairly steady increase in area from 1987 to 2010 equating to an annualized rate of 0.7% over the 23-year period (Table 3.6, Figures 3.4 and 3.5). These findings support other research showing that lake surface area and volumes are increasing concurrently with glacier retreat (Emmer et al. 2014). The Water class decreased 32 hectares between 1987 and 1990 (Figure 3.7) at a rate of -0.4% per annum, but then gained area in all subsequent time steps at rates ranging from 0.5 – 1.6% per annum (Figure 3.5).

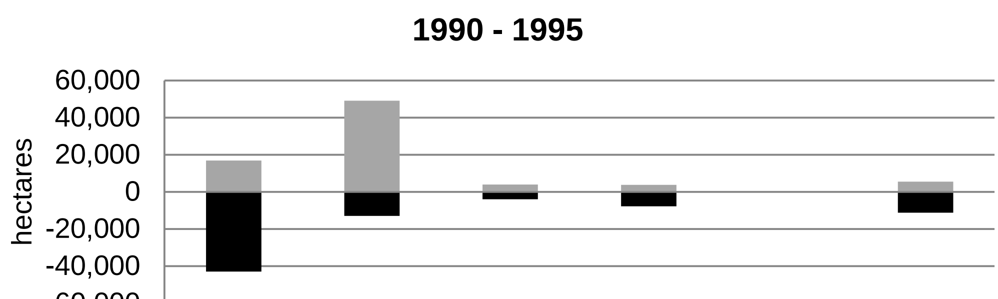


	1987-1990	1990-1995	1995-1999	1999-2005	2005-2010	1987-2010
Barren	17.7%	-6.8%	6.1%	2.4%	-8.0%	0.8%
Puna	-5.8%	4.3%	-1.4%	-2.1%	4.3%	0.3%
Wetland	4.0%	-0.2%	3.1%	-1.7%	-14.6%	-2.6%
Snow/Ice	-4.6%	-1.4%	-0.3%	-1.9%	-1.6%	-1.8%
Water	-0.4%	0.8%	1.6%	0.9%	0.5%	0.7%

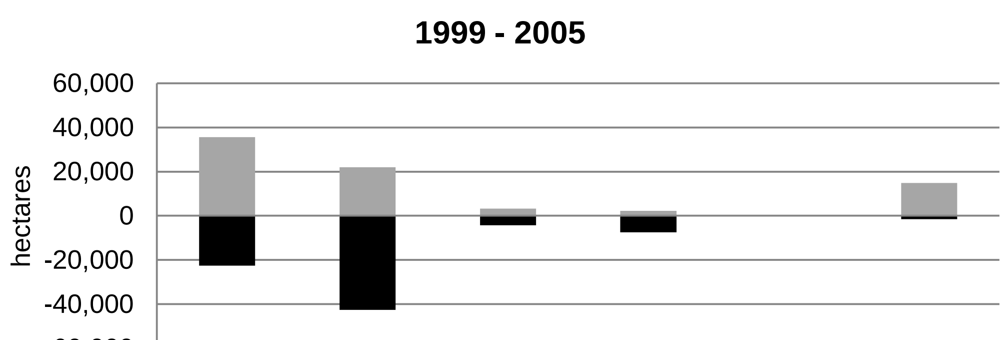
Figure 3.5. Annualized rates of change in area for land cover classes for 5 time steps from 1987-2010 and from 1987-2010.



	Barren	Puna	Wetland	Snow/Ice	Water	Shadow
Gains	50,402	20,405	8,149	2,140	221	9,987
Losses	-13,695	-50,262	-6,625	-11,170	-252	-9,301

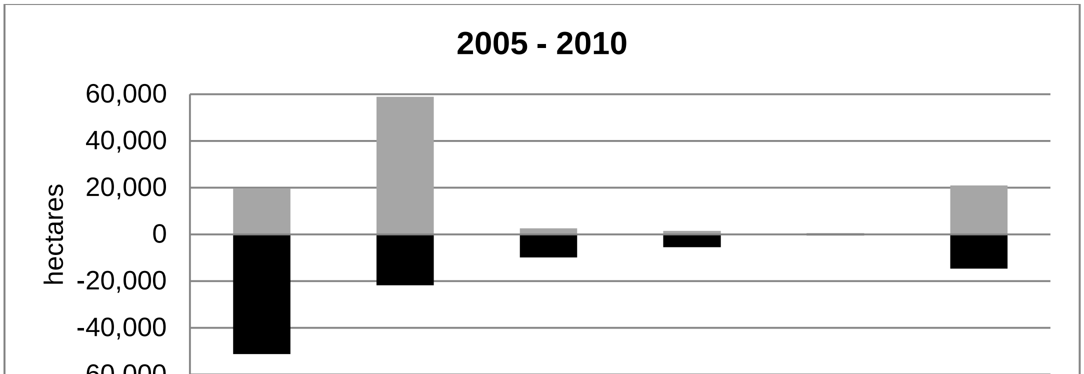


	Barren	Puna	Wetland	Snow/Ic e	Water	Shadow
Gains	16,768	49,087	3,875	3,748	249	5,431
Losses	-42,992	-12,974	-4,015	-7,742	-154	-11,282



	Barren	Puna	Wetland	Snow/Ice	Water	Shadow
Gains	35,568	22,009	3,311	2,284	163	14,866
Losses	-22,515	-42,524	-4,217	-7,444	-22	-1,480

Figure 3.6. Land cover gains and losses for each time step.



	Barren	Puna	Wetland	Snow/Ice	Water	Shadow
Gains	19,561	58,819	2,503	1,433	470	20,872
Losses	-51,337	-21,768	-9,850	-5,584	-405	-14,713

Figure 3.6. Continued. Land cover gains and losses for each time step.

Table 3.7. Net Change (hectares) for each land cover class for each time step (Cloud omitted).

	1987 – 1990	1990 – 1995	1995 – 1999	1999 – 2005	2005 - 2010
Barren	36,707	-26,224	17,952	13,053	-31,776
Puna	-29,856	36,113	-10,316	-20,515	37,051
Wetland	1,525	-140	1,226	-906	-7,347
Snow/Ice	-9,030	-3,994	-604	-5,160	-4,151
Water	-32	96	162	141	65
Shadow	686	-5,851	-8,420	13,386	6,159

Spatio-Temporal Wetland Heterogeneity

Within this context of dynamic land cover change, Wetlands are also undergoing significant transformations with the biggest changes occurring between 2005 and 2010 when Wetland experiences the highest rate of change, area reductions were the largest, and all metrics exhibited notable changes (Figure 3.4, 3.5, 3.6, 3.7-14, Table 3.7). Over the entire study period, 1987 to 2010, Wetland area inside the park boundaries decreased from 12,402 hectares to 6821 hectares, representing a net loss of -45% over 23 years and an annualized rate of change of -2.6%. This rate is faster than the -0.23% decrease in wetlands in the Tibetan Plateau between 1970 and 2006 (Zhao et al. 2015). Changes in area and annualized rates of change were heterogeneous within the 5 change time steps (Tables 3.6, 3.7, Figure 3.5). With the exception of the 1987-1990 period when Wetland increased at a rate of 4.0% per annum, rates of change were negative and ranged from -0.2% to -14.6% (Figure 3.5). Like other land cover classes, Wetland experienced both gains and losses in area over the five time steps (Figure 3.6). Net changes alternated between positive and negative for the first four time steps with a final major net loss of

7347 hectares between 2005 and 2010 (Table 3.7). Between 2005 and 2010, Wetland area decreased by 7347 ha (Table 3.7) at a rate of -14.6% (Figure 3.5).

Landscape ecology metrics that are expressions of spatial pattern and configuration of the Wetland class were computed for each of the time periods (Figures 3.7-3.14; note differing y-axes in all figures). The number of Wetland patches, representing individual wetlands in HNP, was 6912 in 1987, increased to 8766 in 1990, peaked at 8871 in 1999, and declined to 8399 in 2010 (Figure 3.7). Total edge experienced a non-linear trajectory over the 23 years. The biggest increase was from 4377 km in 1987 to 5181 km in 1990, then peaked at 5394 km in 1999, and finally decreased to 3501 km in 2010 (Figure 3.8). Mean patch area was 1.79 ha in 1987, remained stable at ~1.6 ha in 1990 and 1995, then increased to 1.7 ha in 1999 before decreasing steadily to 1.63 ha in 2005 and to 0.81 ha in 2010 (Figure 3.9). As expected the two mean core areas showed a similar change trajectory to mean patch area (Figures 3.10, 3.11).

Mean core area was computed using two different edge distances, 10 m and 50 m, accounting for different depths of edge effects on patches. Only two edge distances were identified in the literature for similar ecosystems. Tovar and others (2012) used a 100 m inside buffer distance for *jalca* (moist Andean grasslands) patches in northern Peru. Kintz and others (2006) used a 30 m edge distance for land cover metrics in a Peruvian national park buffer zone. Because wetlands are known to be long linear patches in valley bottoms with widths usually less than 100 m, I used two buffers that are likely to be better suited for wetlands: 10 and 50 m. In both cases, there was a slight decline from 1987 to 1990, followed by no change or very slight change from 1990 to 1995, a small increase from 1995 to a peak value in 1999. In the final two intervals there was a small decrease from 1999 to 2005 followed by a large decrease from 2005 to 2010 (Figures 3.10, 3.11). More precise buffer lengths would require additional in-depth research to determine the appropriate edge effect specific to high altitude wetlands.

Results shown in Figures 3.9 and 3.10 (Mean Patch Area and Mean Core Area with a 10 meter buffer) are the same because of the method FRAGSTATS uses to calculate core area. The user specifies the edge depth (10 m in this case). FRAGSTATS then places a mask over the cells just outside the patch perimeter. The resolution of the mask is constrained by the cell size (30 m) so the mask is rounded up or down to the nearest cell size given the edge depth (McGarigal 2015). In this case, FRAGSTATS rounds up to 30 m; this is the same calculation as Mean Patch Area that is performed on a 30 m cell size without any edge depth. The 50 m edge depth result shown in Figure 3.11 is different because the mask is rounded up to 60 m.

The Shape Index, which measures the complexity of a patch shape relative to a square, exhibited a decline from 1.26 in 1987 to 1.25 in 1990, was approximately stable until 2005, and then decreased to 1.21 in 2010 (Figure 3.12). The Cohesion metric, a measure of physical connectivity among Wetland patches, was relatively stable from 1987 to 2005, ranging from 92.08 to 92.81 until the largest change occurred from 92.31 in 2005 to 84.07 in 2010 (Figure 3.13). Changes in Nearest Neighbor measured in Euclidean distance from patch edge to patch edge ranged from 4 – 7 m over the study period. This distance is less than the 30 m spatial resolution of the imagery and thus limits its interpretive utility (Figure 3.14).

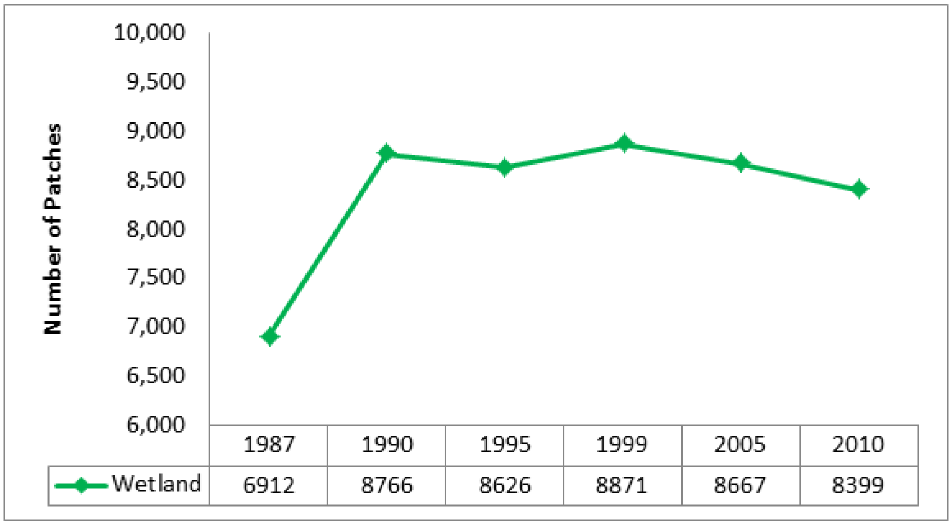


Figure 3.7. Number of Wetland patches over time.

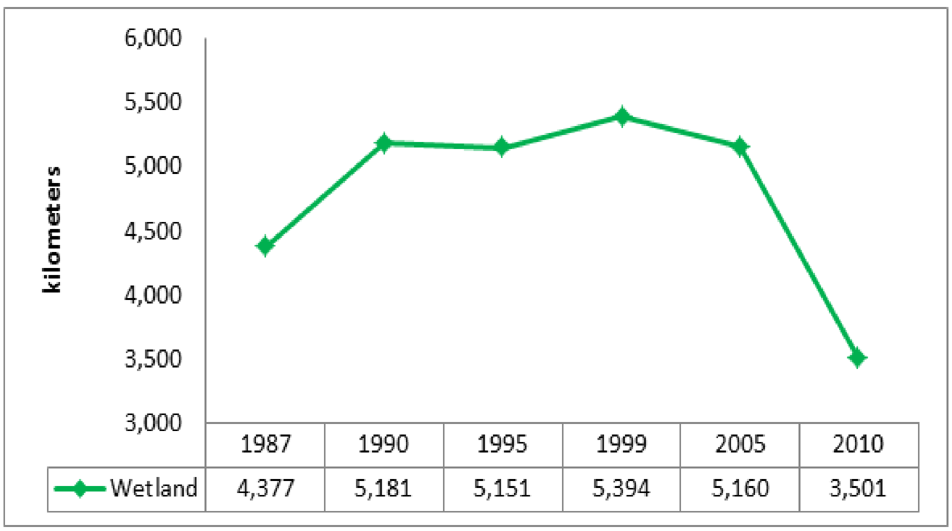


Figure 3.8. Total Wetland edge over time.

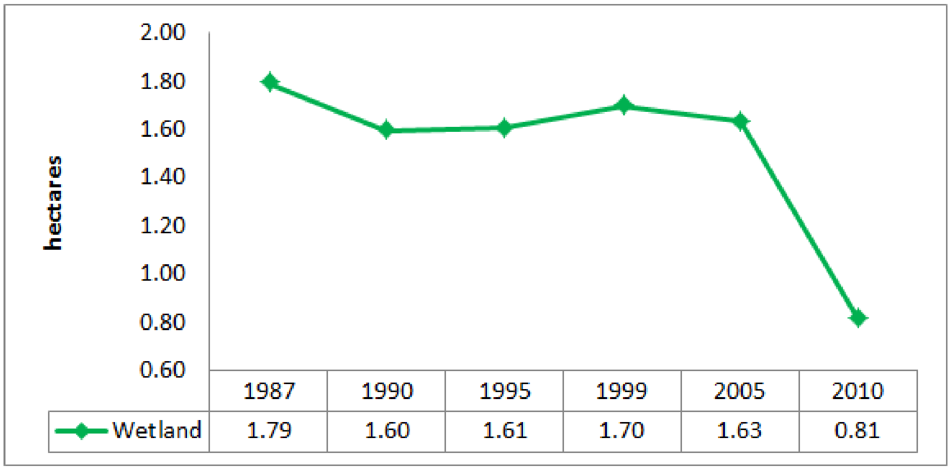


Figure 3.9. Wetland mean patch area over time.

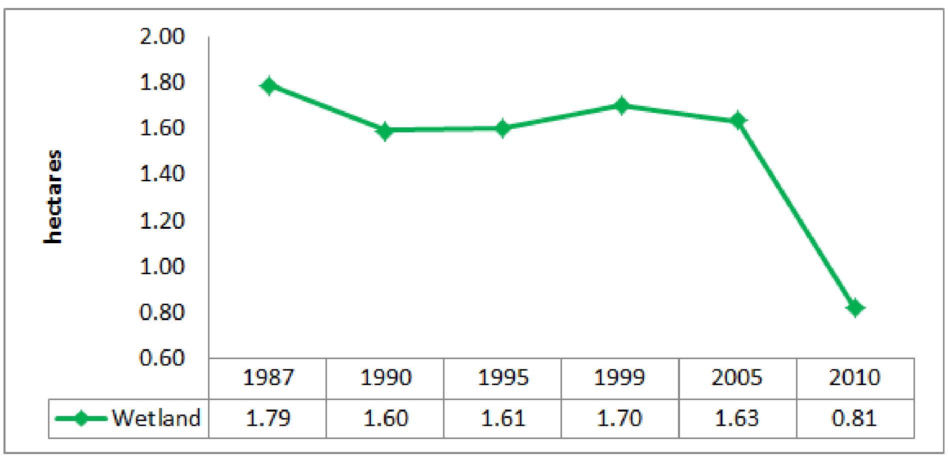


Figure 3.10. Wetland mean core area with 10 meter buffer over time.

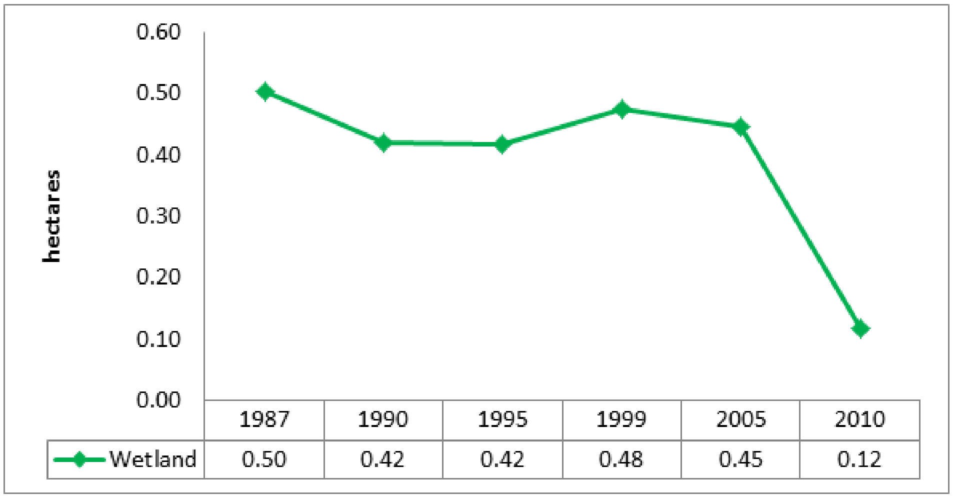


Figure 3.11. Wetland mean core area with 50 meter buffer over time.

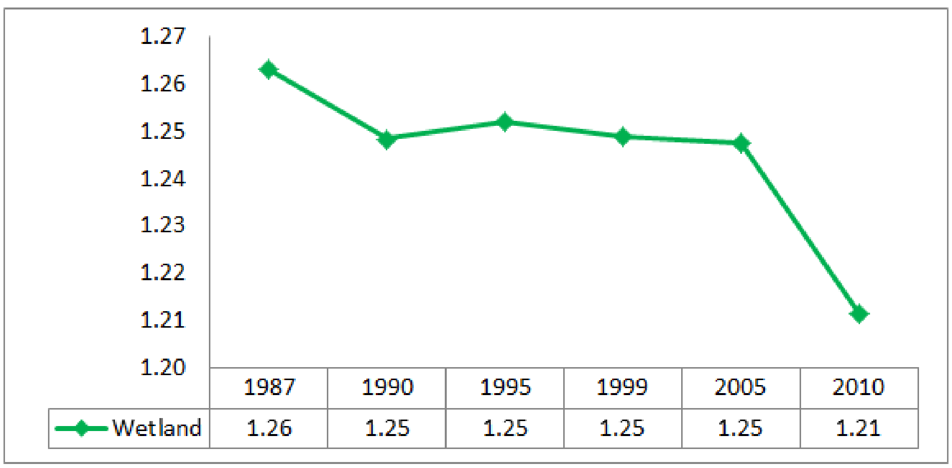


Figure 3.12. Wetland Shape Index over time.

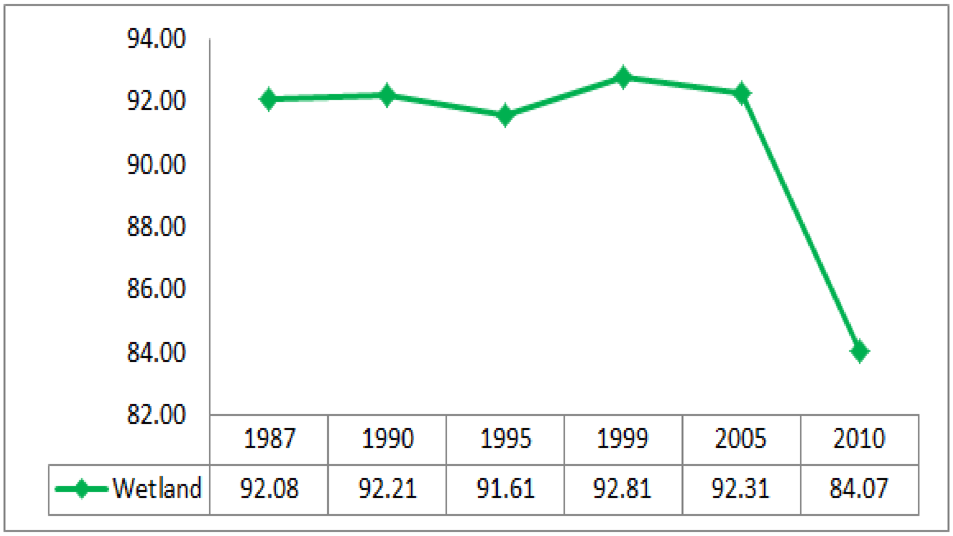


Figure 3.13. Wetland Cohesion over time.

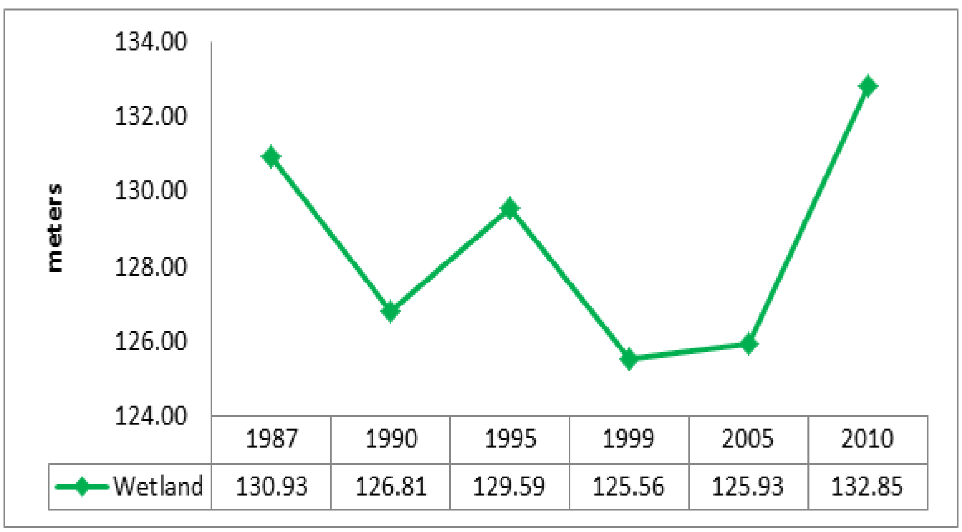


Figure 3.14. Wetland Euclidean Nearest Neighbor over time.

The percentage of landscape occupied by Wetlands over time does not vary widely and suggests that these systems are experiencing stasis (Figure 3.4) despite dramatic shifts in the other land cover classes (Figure 3.2). This finding is in agreement with Dise's conclusion that peatlands exhibit resiliency and are "characterized by quasistable equilibrium states" (2009, 811); however, evidence from other measures illustrate an ecosystem in transition over the 23-year study period. The initial increase in number of patches from 1987 to 1990 represents Wetland fragmentation, followed by a period of quasistable equilibrium to 1999. The decline from 8870 patches in 1999 to 8399 patches in 2010 is indicative of attrition (Figure 3.15). Wetlands could be transitioning to a drier state dominated by grasses (Puna) or they may be eroding away to expose bare substrate (Barren) as I observed in areas subject to heavy grazing and where hiking trails traverse wetlands. Fragmentation is further supported by the increase of total edge (TE) from 1987 to 1999, after which time TE decreases (Figure 3.16). The decline in TE corresponds to Wetland patch attrition from 1999 to 2010; as patches disappear from the landscape, TE decreases concurrently. Mean patch area and mean core areas both show that Wetland patches are, on average, shrinking over the entire study period. After 2005, mean patch area drops by ~50%. While Wetland patches disappear after 1999, those that remain on the landscape are smaller in size and subject to edge effects and associated desiccation of peat soils. The decline of the Shape Index shows that Wetland patch shapes are becoming more regular relative to a square (where a square = 1 in the Shape Index). Wetlands are long and linear on valley floors and as they shrink in size, the decreasing Shape Index indicates that Wetlands are losing area from the ends thus becoming shorter and more square-like. As the length to width ratio decreases, the Shape index approaches 1. The Cohesion metric provides further evidence that Wetland patches are less physically connected over time as attrition takes place and as patches shrink (Figure 3.17). All of these changes are dynamic and nonlinear – annualized rates of change vary over the 5 time steps.

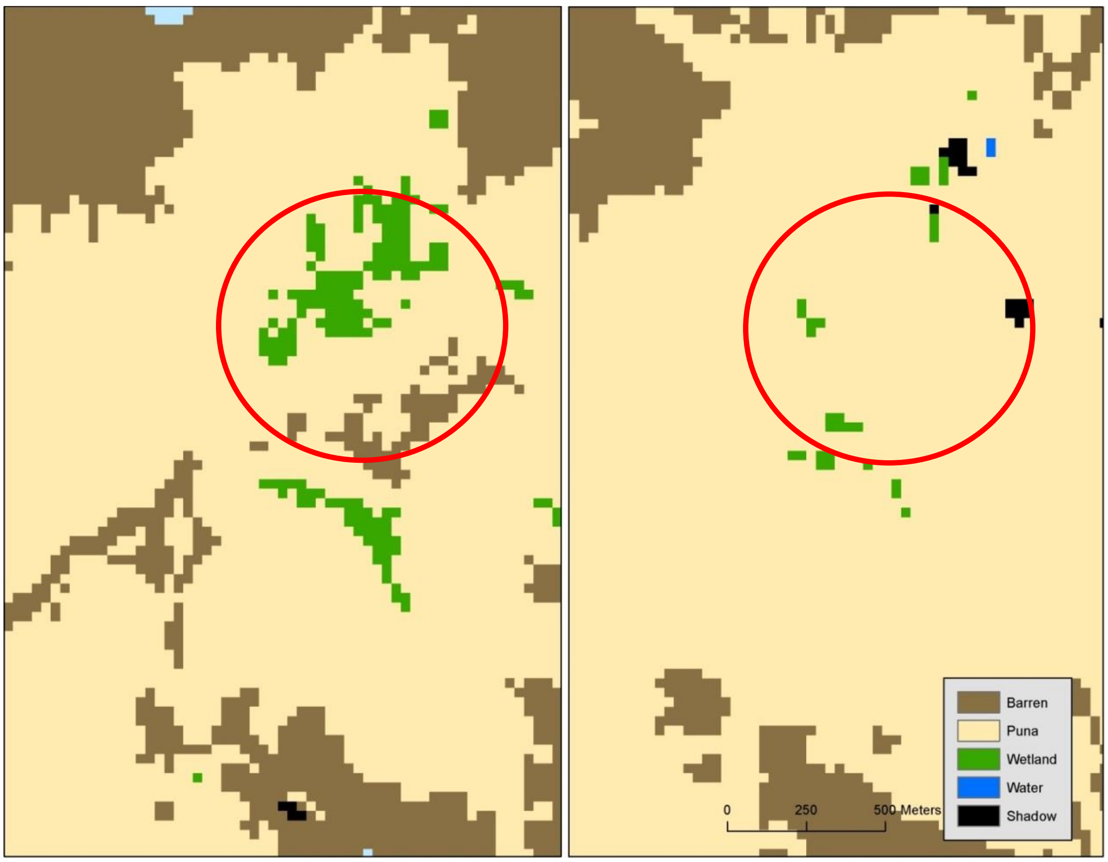


Figure 3.15. Attrition example from 1999 (L) to 2010 (R). Wetland patches in the red circle almost disappear.

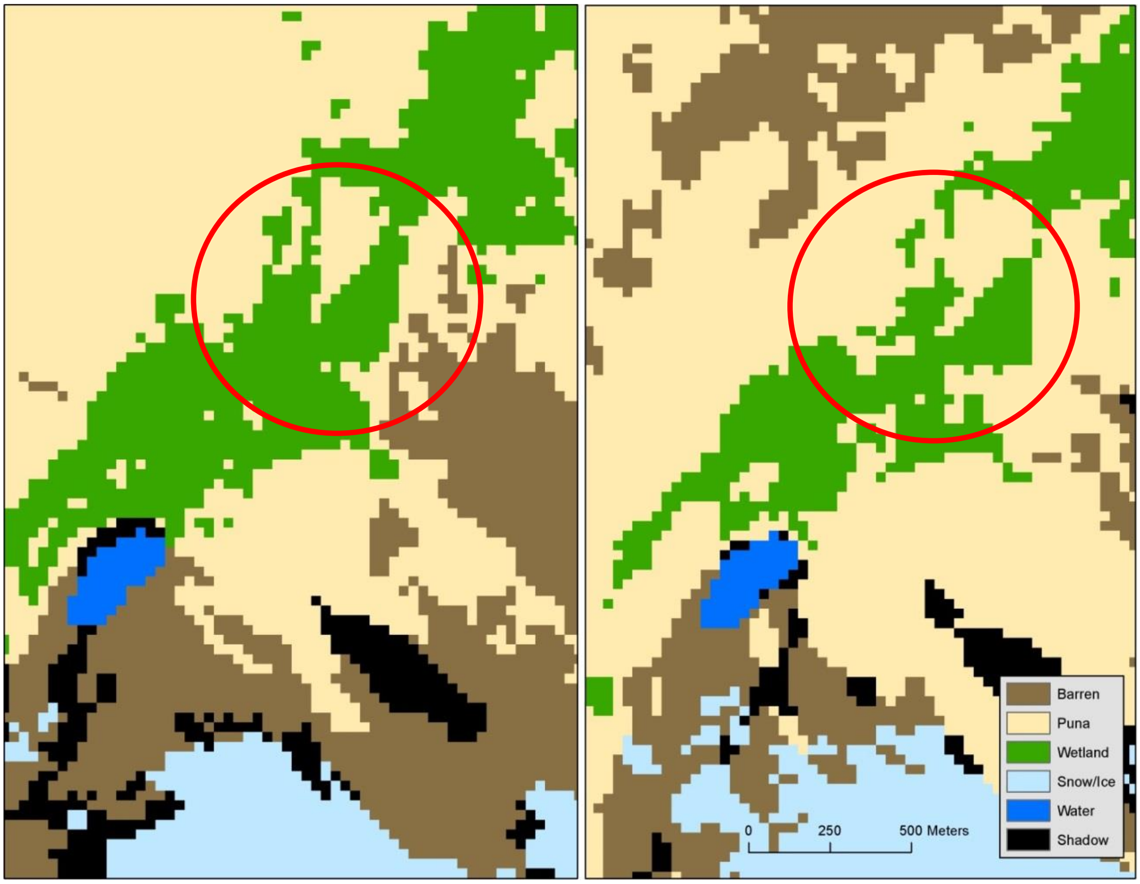


Figure 3.16. Fragmentation example from 1990 (L) to 1995 (R). Wetlands break into smaller patches.

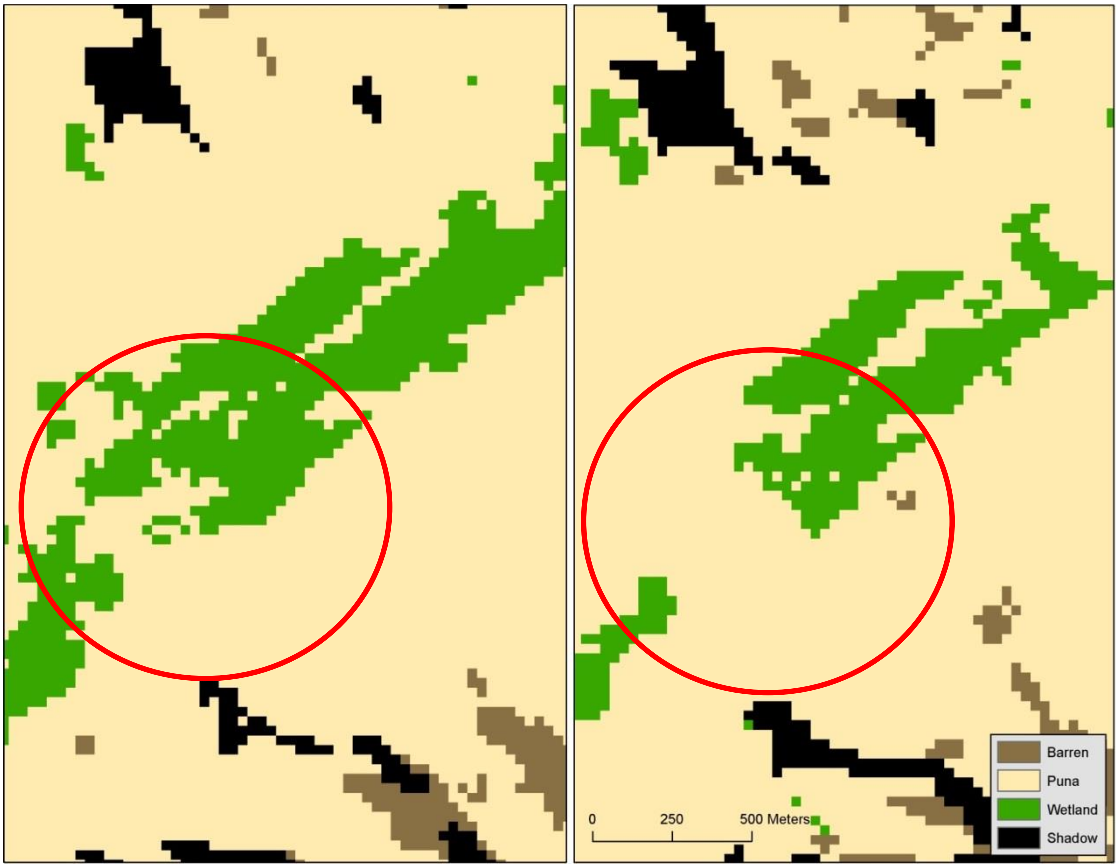


Figure 3.17. Shrinkage and isolation example from 1987 (L) to 1990 (R). Wetlands in the red circle decrease in size and inter-patch distance increases.

During each time step, the individual land cover contributions to net Wetland change vary. These from-to contributions and interactions are clarified in Figure 3.18. The figure illustrates the major Wetland change trajectories and the contributions to net Wetland change expressed in hectares. There were no exchanges between Wetlands and Snow/Ice during any of the intervals so Snow/Ice is omitted from Figure 3.18. From 1987 to 1990, 16 ha of Barren, 823 ha of Puna, and 13 ha Water converted to Wetland. During the next time step, 1990 to 1995, Wetland area decreased, losing 728 ha to Puna. Between 1995 and 1999, Wetland decreased by converting 24 ha to Barren and increased by 814 from Puna. Wetland change from 1999 to 2005 followed a similar pattern, converting 174 to Barren and increasing 1304 from Puna. In the final period from 2005 to 2010, Wetland expanded at the cost of 31 ha of Barren, but decreased by 4251 ha to Puna and 4 ha to Water. Figure 3.18 shows that Wetland spatial changes result from interactions with Barren, Puna, and Water, the three land cover classes that are adjacent to Wetlands, interactions that are addressed in the following section.

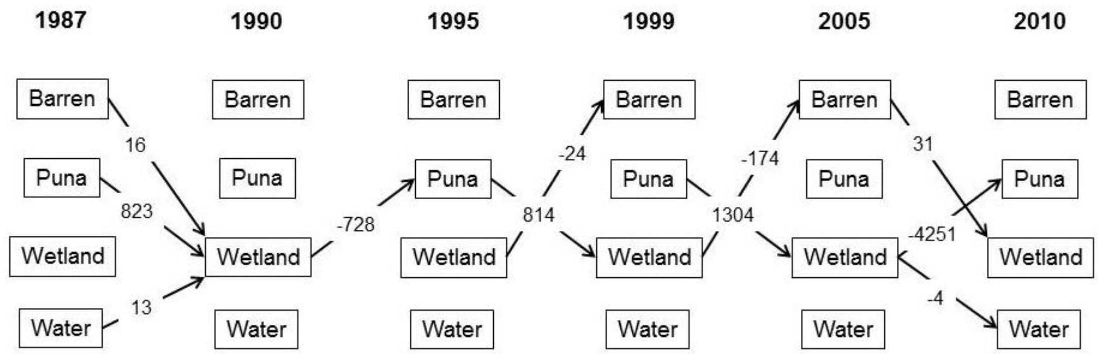


Figure 3.18. Major Wetland change trajectories and contributions to net change shown in hectares. Arrow direction represents direction of change. Negative values represent Wetland loss and positive values represent Wetland gain. Changes > 2 ha are shown. Snow/Ice, Shadow, and Cloud are omitted.

DISCUSSION AND CONCLUSION

While increasing hydrologic variability and associated elevated human vulnerability have received attention in the literature (Bury et al. 2011; Wrathall et al. 2014), this study expands on these glacier recession-related consequences by revealing that landscape dynamism among land cover classes is occurring simultaneously along an elevation gradient. At the highest elevations of the park (>5000 m.a.s.l.), glaciers recede and underlying substrates are exposed, resulting in increasing Barren cover. At the same time, Puna is migrating upslope at the ecotone between Puna and Barren. Upslope plant migration in alpine areas is expected to occur with climate change as warmer temperatures create suitable habitat at higher elevations (Körner 2003; Feeley 2012; Morueta-Holme et al. 2015). Grasses and forbs (mostly Asteraceae) have begun to colonize Barren areas resulting in upslope primary succession (Young 2014), a process observable in the field around 4000 - 4500 m.a.s.l. Rates of change between Puna and

Barren vary from one time step to another, both increasing and decreasing (Figure 3.5). The ecological successional processes explaining this variation are not understood at this time, beyond the scope of this study, and warrant further inquiry.

Water, primarily representing proglacial lakes, occupies <1% of the landscape and increased steadily in area from 1987 to 2010 at an annualized rate of 0.7%. These findings are in agreement with other research showing that lake surface area and volumes are increasing concurrently with glacier retreat (Emmer et al. 2014). As glaciers release water during the melting phase, lakes below the glaciers receive increased input. Increasing lake areas and depths are associated with elevated hazard risk from glacial lake outburst floods (GLOFs) and flows that overtop moraines (Carey 2005, 2010). Changes in Water over the five time steps show that net changes in pro-glacial lake area are the product of both gains and losses. So although the general lake trend is expansion, some lakes may be decreasing. Causes could be attributed to complete loss of ice cover in some valleys, small GLOFs, or intentional lowering of lake levels to minimize GLOF risk. Thirty-five lakes are managed using specially engineered tunnels, drainage canals, and dams to increase lake safety by draining lakes and maintaining lake volumes at appropriately safe levels (Portocarerro 2014). GLOFs and lake overflows are not uncommon. Carey (2010) lists four GLOFs and lake overflows since 1997 that are considered disasters. Other smaller events also occur. For example, large torrrents of water were released from Lake Palcacocha, Lake 513, and Lake Artizon Alto in 2003, 2010, and 2012 respectively (Emmer et al. 2014; Portocarerro 2014).

An examination of major Wetland change trajectories (Figure 3.18) reveals that Wetland is coupled with the dynamics of Puna, Barren and, to a lesser extent, Water. The interaction between Wetland and Water most likely occurs in the littoral zone of glacial lakes. As lakes gain and lose surface area, although small, wetlands are either inundated by lake water or lakes recede to allow marshes and shore fens to develop. Shoreline wetlands have been observed in the field and are often colonized by *Juncus arcticus var. andicola*, an aquatic rush typical of the Andean highlands (Figure 3.19) (León and Young 1996). The biggest changes overall occurred between Wetland and Puna from 2005 to 2010 when 4251 ha of Wetland converted to Puna. The reason for the timing and magnitude of this change is unknown, but it is a signal that the matrix is expanding. As Wetlands disappear and shrink, surrounding grasslands and shrubs would be expected to invade previously waterlogged areas. Future research that incorporates 2015 and 2020 imagery (and beyond) could assist in describing the temporal context of Puna expansion and would offer insight into the timing of these vegetation shifts.

Figure 3.19. An example of shoreline wetlands dominated by *Juncus arcticus* var. *andicola* in the Pastoruri valley.

Interactions between Barren and Wetland are larger in magnitude than Water and Wetland (Figure 3.18). The conversion from Barren to Wetland 1987 to 1990 and 2005 to 2010 could result from primary peat formation, which occurs when peat soils expand over adjacent substrates. It is known to take place in glacial forelands following ice loss (Rydin and Jeglum 2013) (Figure 3.20). Lateral peat growth over adjacent wet mineral soils or geologic substrate would account for conversions from Barren to Wetland, but additional field observations over time would be required to confirm this explanation.

Figure 3.20. Formation of new peat over a rocky substrate that was covered by a glacier in the Pastoruri valley.

A possible explanation for the conversion from Wetland to Barren is erosion. Wetlands are susceptible to erosion when hydrologic connectivity is disrupted and when disturbances trigger erosion (Charman et al. 2008). Changes from Wetland to Barren could occur when water inflows are disrupted. Loss of hydrologic input interrupts the positive water balance necessary to maintain saturated soil conditions and leads to drier soils that are then highly susceptible to erosion. Local informants report that springs often found in wetlands in HNP are disappearing (see also Mark et al. 2010; Baraer et al. 2014), a sign that loss of hydrologic input to wetlands may be taking place. Erosion could also be triggered by trampling, kicking, and grazing by cattle and horses. The activities of these animals increase soil erosion when 1) soil compaction leads to diminished water absorption; 2) hooves loosen soil mechanically; and 3) destruction of dense vegetative cover makes soil vulnerable to erosion (Millones 1982). Similar phenomena have been described in the montane soils in Venezuela where cattle exfoliate turf (Pérez 1992a, 1993) and were observed *in situ* (Figure 3.21). The round depression, or pan, shown in Figure 3.21 is similar to others observed in the Venezuelan páramo (Pérez 1992b) that tend to deepen and widen over time. In addition to cattle as agents of erosion, pans in the Venezuelan páramo are eroded by needle ice activity, surface wash, and throughflow thus exposing underlying coarse glaciofluvial sediments (Pérez 1992b). Erosion can also take place during the rainy season when stream discharge is at its highest. Lateral bed movement across valley floors erodes wetlands by washing away wetland soils and, in turn, exposing underlying gravel that would be classified as Barren. Losses of Wetland to Barren are of concern because erosion of peat soils mobilizes stored carbon, nitrogen, phosphorus, and sulfur. Erosion impedes the ability of wetlands to sequester carbon (Strack 2008) and the accompanying disruption of ecosystem integrity threatens biodiversity.

Figure 3.21. A depression approximately 2.5 meters in diameter created by cattle digging in peatlands. Image taken in the Quilcayhuanca valley.

The interactions highlighted by Figure 3.18 uncover a close coupling between Puna and Wetlands. During the time steps 1987-1990, 1995-1999, and 1999-2005, a total of 2941 ha of Puna converted to Wetland. From 1990-1995 and 2005-2010, 4979 ha of Wetland converted to Puna. Although there are 3 periods when Wetlands gain area, total losses to Puna outweigh gains over the 23-year period. The precise mechanisms responsible for Wetland gains relative to Puna is unknown, but may be related to springs in talus deposits. During years with increased precipitation, flow from these springs increases temporarily. Elevated hydrological input into wetlands could explain Wetland gains relative to Puna. Processes driving Wetland losses to Puna could be a combination of decreased discharge related to ice loss, disappearance of springs, as well as overgrazing, trekking traffic, camping, and ditch construction and maintenance. These disturbances result in Wetlands that are changing spatially through fragmentation, attrition, and increased isolation among patches.

Relative to the four possible alternatives outlined in the chapter Introduction, the final alternative – nonlinear change and a combination of transformation processes – most closely characterizes spatio-temporal Wetland change observed in these data. Over the 23-year period, Wetland spatial distribution is described by increasing patch-to-patch isolation, fragmentation, shrinkage, and attrition. Three Wetland metrics exhibited nonlinearity: Wetland area, number of patches, and total edge (Figure 3.6, Figures 4.7, 4.8). In catchments that are losing ice cover, stream discharge responds to glacier recession in a nonlinear fashion (Barnett, Adam, and Lettenmaier 2005; Huss et al. 2008; Baraer et al. 2012). A logical follow-up question to this chapter is: What is driving Wetland change? It is possible that a statistical relationship between discharge trends and Wetland spatial dynamism could be established and might pinpoint causal mechanisms. Any research on causality should also include an examination of variables besides stream discharge. This question is further explored in the following chapter.

If patch-to-patch isolation, fragmentation, shrinkage, and attrition continue into the future, the ecological effects of wetland changes will likely be significant. If these trends are accompanied by increasing temperatures and further interruptions of hydrologic connectivity, then ecological effects will amplified. Negative ecological consequences could include losses of biodiversity and plant and animal assemblies (Young 2014). Charman and others (2008) described the projected ecological effects of climate change on peatlands. Vegetation that forms dense mats on peat surfaces and adapted obligate species are not expected to survive warmer, drier conditions. Loss of peat vegetation would expose underlying soils and make them vulnerable to desiccation. Peat soils that dry out experience structural changes that lead to accelerated erosion stored carbon can be mobilized. Water storage capacity would be degraded thus enhancing downstream flooding (Charman et al. 2008). That peatlands in HNP will likely mirror Charman's (2008) projected impacts of climate change on peatlands is a reasonable assumption.

Along with these predictions, I offer cautionary notes and ideas for future research directions to improve the detection of peatlands in HNP. Landscape metrics are an acceptable method for characterizing landscape structure and used widely, yet they remain imperfect. They are sensitive to grain, study area extent, and thematic resolution; demonstrating that spatial pattern change represents ecological change remains a challenge and new approaches have been proposed (Kupfer 2012; Turner and Gardner 2015). Seven land cover classes were defined based on other land cover change studies in the same study area (Lipton 2008; Silverio and Jaquet 2009) to permit comparisons in future work. Liu and colleagues (2013) demonstrated that landscape classification schemes influenced most of the metrics they selected in magnitude, but in ways that differed when comparing smaller to larger extents. Huang and colleagues (2006) showed that when using fewer than ten land cover classes in extents <100 km^2, metrics fall into a "sensitivity window" where adding or removing a class could result in significant differences in the metric results. In this case, the study area is 3400 km^2, well above the extent noted by Huang and colleagues (2006), but given the sensitivity of metrics to thematic resolution, future research could generate a finer classification scheme (more than seven classes). The revised classification scheme would ideally differentiate between permanently wet peatlands and seasonally wet meadows, and potentially the 5 vegetation communities described in Chapter 5. Capitalizing on technologies such as field spectroradiometers and fusing data from optical and active sensors (see for example Bourgeau-Chavez et al. 2015), future work would ideally classify all peatlands in the study area in finer spatial and thematic resolution, although the temporal depth provided by Landsat TM would be compromised. Future work would also explore quantitatively the ecological significance implied by changing metrics and a test of statistical significance would be useful to determine the changes from one time period to another.

Keeping the aforementioned cautionary statements in mind, the predictions for peatlands nevertheless forefront the importance of developing strategies to manage anthropogenic disturbances and two new projects provide an appropriate forum. Officials have commenced writing the next park master plan (Gómez 2014) and this version offers an opportunity to prioritize wetlands and their ecosystem services. The goal of a new restoration project organized by park officials, The Mountain Institute, and the U.S. Forest Service is to establish core knowledge for wetland restoration action and monitoring that can be applied park-wide (O’Donnell 2015). Both of these activities could readily address wetland threats, especially overgrazing. Using a social-ecological framework, agronomists could work in conjunction with communities and government officials to develop sustainable grazing strategies (Polk and Young In press). Trekkers need to be constrained to a single track and camping should be banned on wetlands or near susceptible wetland edges. Community-initiated ditch construction and maintenance for the purpose of improving pasture could also be limited, although the impacts to livelihoods should be evaluated prior to implementing any new management practices (refer to Chapter 6 that discusses pastoralist uses and benefits from peatlands through their livestock). Natural disturbances are difficult and perhaps impossible to manage. Decision makers will have to develop and implement strategies that benefit and protect transforming wetlands in a tropical high altitude landscape.

Chapter 4: Tropical Mountain Wetland Change, Hydrologic Processes, and Climate

INTRODUCTION

The connections between land change and tropical hydrologic processes are not well understood, prompting calls for more research in this area (Wohl et al. 2012). In practice, quantifying the relationship between land change and hydrologic processes remains challenging for several reasons, as noted by DeFries and Eshelman (2004). Hydrological records for systems with high natural variability are often incomplete or short-term. Designing *in situ* experiments and sampling strategies is impeded by the inability to control variables in remote locales. Extrapolating to other systems is cautionary at best. Now, with satellite data widely and freely available and advances in modeling capabilities, understanding the interactions between land change and hydrologic processes is more feasible (DeFries and Eshleman 2004). In this chapter I apply an econometric model to explicate the relationship between changing wetland area and hydrologic processes and climate variables in a high elevation tropical landscape characterized by glacier recession. By doing so, the analysis shows that wetlands are connected hydrologically to melting glaciers and related discharge. The approach presented could be applied in other rapidly changing glaciated catchments in the tropics in order to better evaluate the links between landscape change and hydrologic processes.

In this chapter, I follow Pringle's definition of hydrologic connectivity: "water-mediated transfer of matter, energy and/or organisms within or between elements of the hydrologic cycle" (2003, 2685). Using in-stream instrumentation and hydrological models in Cordillera Blanca watersheds, researchers are making important advances regarding surface and groundwater hydrologic processes and connectivity during a time of accelerating glacier recession (Baraer et al. 2012, 2014; Carey et al. 2014; Gordon et al. 2015). As mentioned earlier in this dissertation, the area is also a research hub for tropical glacier recession and associated social and biophysical impacts (Mark and Seltzer 2003; Juen, Kaser, and Georges 2007; Baraer et al. 2009; Mark et al. 2010; Chevallier et al. 2011; Lynch 2012; Iturrizaga 2014; Schauwecker et al. 2014). Given that the Cordillera Blanca mountain range is oriented on a northwest-southeast axis and is comprised of a series of parallel valleys, each its own watershed in the Santa River basin, the arrangement of these watersheds and their varying environmental characteristics provide a living laboratory where watersheds can serve as sample units or watersheds in analyses, particularly regarding the relationships between hydrologic connectivity and landscape change. Land change in the Cordillera Blanca is driven by glacier recession, human activity and socio-ecological feedbacks characteristic of the Andean landscape (Young 2008, 2014). Unlike responses to land change in temperate zones that tend to be more uniform, land change in the high elevation tropics triggers heterogeneous hydrological responses, some of which may be unexpected (Ponette-González et al. 2014). Ecological integrity of landscapes is dependent upon hydrologic connectivity and alterations to it can negatively affect ecosystem functionality (Freeman, Pringle, and Jackson 2007). In the present chapter, I capitalize on a historic discharge dataset from the Cordillera Blanca and combine it with remote sensing and climate data to identify the connections between hydrologic processes and changing wetland ecosystems.

Land change scientists use varied epistemological approaches to attempt to identify causal factors and draw on backgrounds including remote sensing, political ecology, resource economics, institutional governance, landscape ecology, and biogeography, among others. It is an interdisciplinary field that joins human behavior, environmental principles, GIS, and remote sensing (Turner II, Lambin, and Reenberg 2007). Econometric theory is one way to understand forces driving landscape change. Before economists were using spatially explicit approaches, geographers and sociologists innovatively applied econometrics to model the empirical relationships between deforestation and independent variables (Allen and Barnes 1985; Walker 2004). In joining econometric modeling with spatial data to pinpoint forces of land transitions, independent variables can be conceived of as spatial driving forces (SDFs) (sensu Chowdhury 2006). The essence of the SDF approach is to use classified satellite imagery to extract dependent and independent variables used in econometric analysis, thus generating spatially explicit inputs derived from thematic data. Ancillary datasets representing variables such as precipitation, evapotranspiration, geology, topography, population densities, to name a few, can further specify the model. These spatially explicit inputs can be combined with spatial biophysical and social data collected in the field. Thus, when econometric modelling is applied to land change science, the epistemology is quantitative and spatially explicit using georeferenced data (Chowdhury 2006).

A commonly used econometric technique utilized by land change scientists is multiple logistic regression, or logit models (Mertens and Lambin 2000; Serneels and

Lambin 2001; Serra, Pons, and Saurí 2008; Wyman and Stein 2010; Schulz et al. 2011; Cui et al. 2014). Logit models provide a probability that estimates the effect of the independent variables on the dependent variable (Wooldridge 2009). When used in land change science, logit models estimate the probability that processes like deforestation, agricultural expansion, or wetland loss will occur. In these cases, the dependent variable represents the probability of a particular land cover presence (or absence) occurring at a given point in time (the image acquisition date). Land change scientists may apply the logit model to panel data, or two or more classified images from two or more different dates. Often the dependent variable equals 0 when there is no change from Time 1 to Time 2 and equals 1 when there is a change detected between Time 1 and Time 2. The analyst can then predict the probability of a land change in the future (Chowdhury 2006). When the estimated probability lies between 0 and 1, the analyst must determine a threshold value where change is expected to occur. For example, if the estimated probability is 0.7, does that indicate that change is likely? So although the predictive capacity of logit models is attractive, interpretation remains difficult (Wooldridge 2009) because determining the threshold value between 0 and 1 where land change will occur is a subjective decision made by the researcher.

Another technique for modeling land change using econometrics involves panel data and fixed effects regression (FE). Economists, psychologists, political scientists, and sociologists have long used longitudinal panel data and FE in their research (Andreß, Golsch, and Schmidt 2012). Typically the unit of observation is an individual, household, firm, or country (Allison 2009). When two time periods are used, the technique is referred to as first differencing (FD) (Wooldridge 2009). Sociologists have used FE to evaluate environmental change as it relates to shifts in population, economic forces and other factors. For example, Jorgenson and Clark (2011) used FD on the country level to identify the relationship between per capita ecological footprints and consumption-based economic metrics from 1960 to 2003. Clement and Podowski (2013), used FD to show that loss of cropland to the built environment from 2001 to 2006 has a stronger statistical association with net migration rates than population growth. They also found that increasing incomes are negatively associated with land intensification. Clement and others (2014) used FD to show that increasing rural populations may be associated with afforestation in the southern United States, an unexpected finding that questions the notion that population growth always causes deforestation. What differentiates the present study from others is that the dependent and independent variables are strictly biophysical and I empirically determine the association between wetland change and hydrology as it is moderated by global environmental change.

The FE technique relies on a simple design: make each observation its own control, as is the case in this research where each sampled watershed is an observation. This is useful in studies where a non-experimental design is the only available strategy to evaluate causal inferences. The dependent variable in FE methods must be measured on two occasions at a minimum and these measurements must be directly comparable over time. Additionally, the independent variables must exhibit some variation across time (Allison 2009). In contrast to land change logit models that measure a probability response for change from one land cover to another, the fixed effects model generates an estimate of the relative effect of an independent variable over time. It explicitly estimates how changes in the independent variables affect change in the dependent variable in the temporal dimension (Wooldridge 2009). In most regression models, including logit models, endogeneity is strict: independent variables should not correlate with the error term. An important aspect that differentiates FE from other models is that the endogeneity assumption is relaxed. Independent variables are permitted to correlate with unobserved time-constant variables and the error term, together referred to as the composite error (Wooldridge 2009). Relaxing endogeneity is a crucially important deviation that is particularly relevant to land change studies because some independent variables may not be observable and are time-constant, yet, along with random error, they can together affect the dependent variable. When unobserved variables are allowed to have any association with the observed variables, as in the case of FE, the model can then control for unobserved time-constant variables (Allison 2009). Mathematically, the FE technique subtracts away the time-constant variables and the error term. Doing so eliminates time-constant effects, both observed and unobserved (Andreß, Golsch, and Schmidt 2012). If the goal is to identify spatial driving forces of a changing dependent variable, then independent variables that are not changing cannot explain change in the dependent variable.

Like other glaciated tropical mountains, the Cordillera Blanca is undergoing glacier loss (Kaser 1999; Thompson et al. 2006; Rabatel et al. 2013). In the 1930s, glacier area was 800-850 km^2 (Georges 2004) and declined to 482 km^2 by 2010 (Burns and Nolin 2014). The relationship between glacier loss and streamflow is thought to be nonlinear in the Cordillera Blanca (Baraer et al. 2012) and fits the expectations of Jansson et al. (2003). While ice cover steadily declines over time, stream discharge first increases responding to an initial pulse of meltwater. Then, discharge reaches an apex and begins to decrease because the meltwater produced by the glacier decreases as mass and cover decrease. Consequently, annual and dry-season discharges decrease and annual variability increases. Downstream, these shifts in water availability have serious consequences for human consumption, agriculture, and ecosystem integrity that depend on streamflow sourced from tropical glaciers (Bradley et al. 2006). In the Cordillera Blanca, decreasing streamflow and reliability have been linked to increased human vulnerability and social conflict, and changing ecosystems, among other consequences (Bury et al. 2013; Carey et al. 2014).

Within the valley mosaics of the Cordillera Blanca, peatlands and wet meadows (together referred to hereafter as wetlands) form patches that are becoming smaller and more fragmented as shown in the previous chapter and elsewhere (Bury et al. 2013). Identified in Spanish as *bofedales* (Squeo et al. 2006; Cooper et al. 2010) or *pampas* (Mark and McKenzie 2007; Baraer et al. 2014), they are a component in hydrologic connectivity, interacting with groundwater and streamflow heterogeneously along elevation gradients (Gordon et al. 2015). Wetlands receive input from groundwater, precipitation, and surface water flows (Young 2014) and they contribute to aquifers and streamflow (Gordon et al. 2015). Water movement is thus vertical and lateral through the wetland. Hydrologic outflows from peatlands are seasonally variable, increasing during rainy periods and decreasing in the dry season when evapotranspiration is higher (Rydin and Jeglum 2013). High altitude wetlands are thought to store groundwater that is released during the dry season (Buytaert, Cuesta-Camacho, and Tobón 2011), but they may in fact be only minor contributors to dry season discharge (Baraer et al. 2014; Gordon et al. 2015). Rather than focus on seasonal differences, I explore the relationships between spatially changing wetlands and shifting hydrologic processes over a longer term. To do so, I tested 7 hypotheses (Table 4.1 and illustrated in Figure 4.1).

Table 4.1. Hypotheses exploring the relationships among changing wetlands and hydrologic variables.

H1	Glacier recession has an effect on wetland area; $\Delta\beta_{\text{Glacier\_area}} \neq 0$.
H2	Glacier recession and changing lake area have an effect on wetland area; $\Delta\beta_{\text{Glacier\_area}} \neq 0$ and $\Delta\beta_{\text{Lake\_area}} \neq 0$.
H3	Glacier recession and decreased discharge have an effect on wetland area; $\Delta\beta_{\text{Glacier\_area}} \neq 0$ and $\Delta\beta_{\text{discharge}} \neq 0$.
H4	Glacier recession, decreased discharge, and change in lake area have an effect on wetland area; $\Delta\beta_{\text{Glacier\_area}} \neq 0$, $\Delta\beta_{\text{discharge}} \neq 0$, $\Delta\beta_{\text{Lake\_area}} \neq 0$.
H5	Glacier recession, decreased discharge, and change in prior precipitation have an effect on wetland area; $\Delta\beta_{Glacier\_area} \neq 0$, $\Delta\beta_{discharge} \neq 0$, $\Delta\beta_{ppt} \neq 0$.
H6	Glacier recession, decreased discharge, change in lake area, and change in prior precipitation have an effect on wetland area; $\Delta\beta_{Glacier\_area} \neq 0$, $\Delta\beta_{discharge} \neq 0$, $\Delta\beta_{lake\_area} \neq 0$, $\Delta\beta_{ppt} \neq 0$.
H7	Decreased discharge and change in prior precipitation have an effect on wetland area; $\Delta\beta_{discharge} \neq 0$, $\Delta\beta_{ppt} \neq 0$.

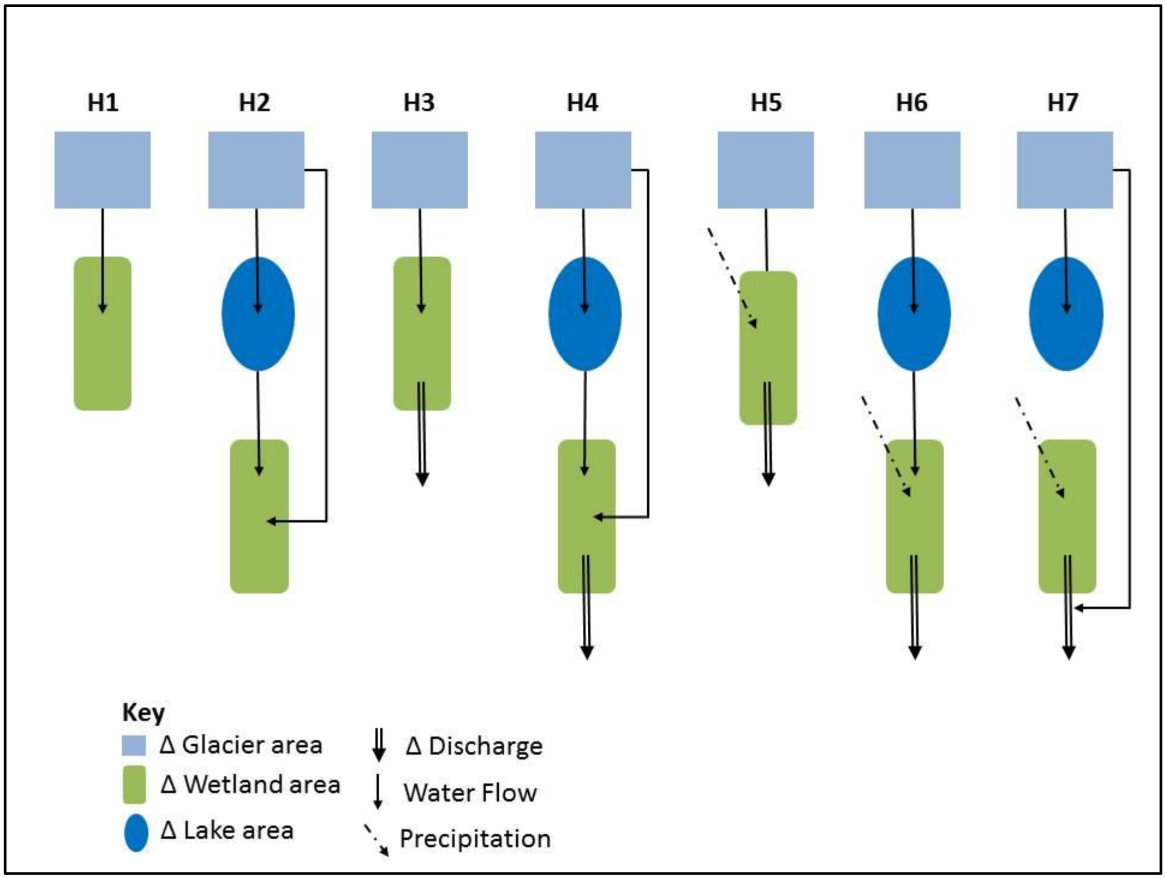


Figure 4.1. Diagram of hypotheses in Table 4.1 exploring the relationships among changing wetlands and hydrologic variables.

Because wetlands are spatially proximal to glaciers, I first hypothesize that glacier area change should be associated with wetland change (H1). Glacier recession is a driver of ecological change in high elevation watersheds (Milner, Brown, and Hannah 2009; Jacobsen et al. 2012), thus wetland ecosystems should also be impacted by these upper watershed dynamics. In most Cordillera Blanca watersheds, proglacial lakes dammed by moraines (or engineered impoundments) store glacier meltwater. Ice loss is associated with increasing lake surface area and volume (Emmer et al. 2014), so wetlands could be changing as a function of both glaciers and lakes (H2).

Increasing lake area and volume provides added water storage that feeds wetlands. Baraer and others (2012) have shown that Santa River discharge is decreasing while variability is increasing as glaciers recede, so I further hypothesize that changes in discharge and glacier area controls wetland area change (H3). Since wetlands are a component of watershed hydrology (Gordon et al. 2015), I expect that changing discharge may be linked to wetland change. Next, I combined H1, H2, and H3 to include glacier area, lake area, and discharge as predictor variables in changing wetland area (H4). In northern Peru, most high altitude peatlands are groundwater supported fens (Cooper et al. 2010); however, based on our field observations some wetlands are perched above the watershed floor and may be cut off from groundwater sources. In this case, these bogs may be supported only by precipitation so H5 introduces a lagged precipitation variable in place of lake area and maintains the relationship between glacier area and discharge. I use a lagged variable because some water held in wetlands may be stored from prior rainy seasons, although the residence time is unknown (Gordon et al. 2015). In H6, I include 4 variables, changes in glacier area, discharge, lake area, and precipitation, to estimate wetland change.

Finally, I consider that change in glacier area may not have any statistically significant association with changing wetlands; instead, decreased discharge and lagged precipitation may alone predict wetland change (H7). Gordon and others (2015) have shown that mid-watershed wetlands depend on precipitation-derived groundwater and interactions with aquifers. As glaciers recede, the influence of meltwater will diminish and precipitation will be the key streamflow driver (Mark and Seltzer 2003; Baraer et al.

2012). Given the paucity of hydrological and meteorological instrumentation in the Cordillera Blanca (Mark, McKenzie, and Gómez 2005), it is difficult to evaluate the relative effect of spatial driving forces on changes in wetland area, but FE offers an option to test the 7 hypotheses and provide insight into understanding hydrologic connectivity of changing wetlands.

METHODS

The scale of analysis is the watershed level; I evaluated 7 watersheds within the Santa River basin (Figure 4.2). The nearby city of Huaraz has approximately 150,000 inhabitants. All sample watersheds are located on the west side of the Cordillera Blanca and drain into the Santa River which flows to the Pacific Ocean. Watersheds on the east side were not used because there are no available discharge records. The 7 watersheds are representative and span the latitudinal gradient of the range, varying in size, glacier coverage, and wetland coverage. The time periods are 1987 and 1995, years in which imagery, climate, and discharge data are available and gap-free. Wetlands occur at elevations > 3500 m.a.s.l. and are protected inside Huascarán National Park, a 3400 km^2 UNESCO Biosphere Reserve. Based on field observations, wetlands in the study area include both highly organic peatlands and minerogenous wet meadows.

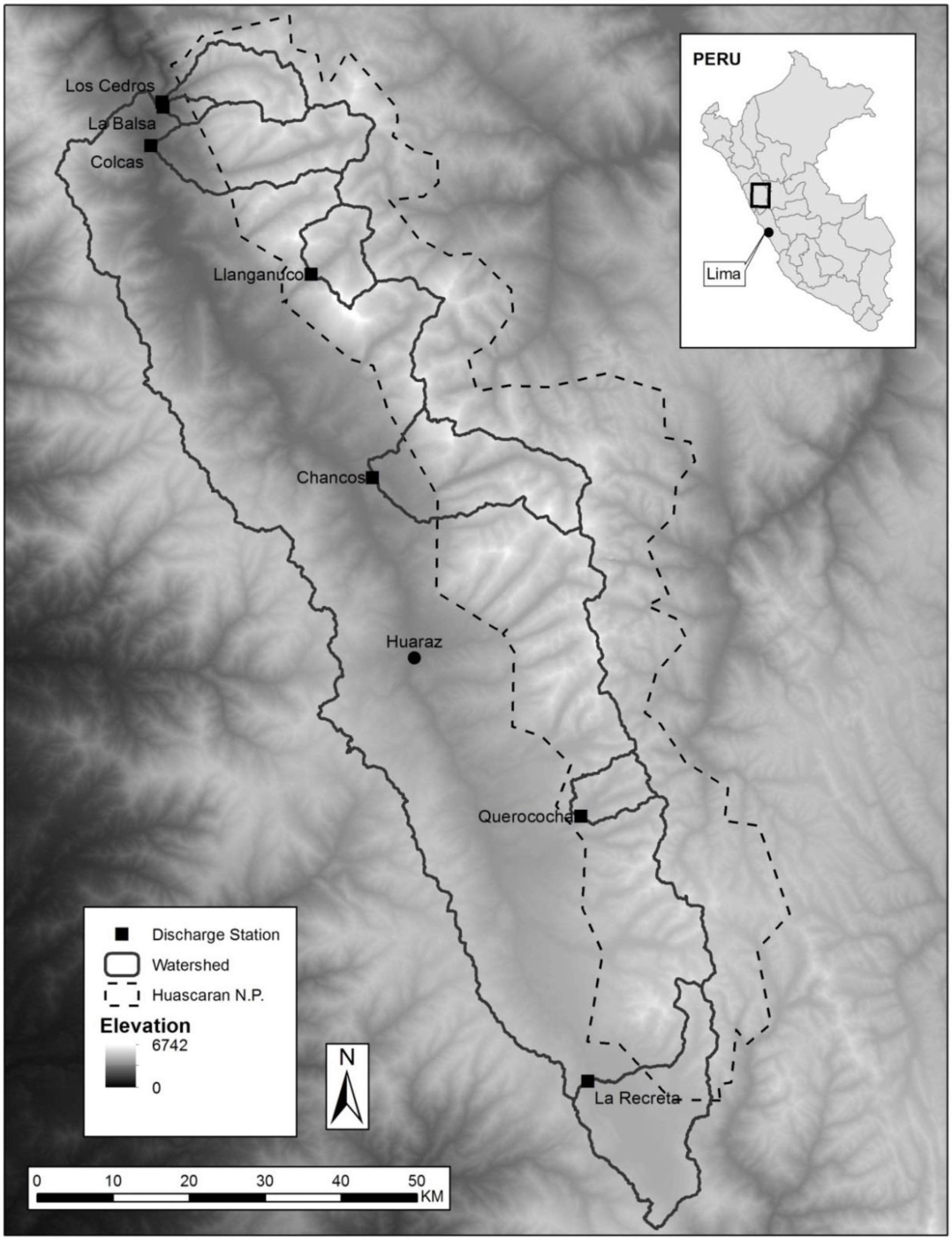


Figure 4.2. Study area showing 7 watersheds, Huascarán National Park and proximity to city of Huaraz.

Methods involved a two-stage process beginning first with extracting the dependent and independent variables (Table 4.2; summary statistics in Table 4.3) followed by the FD regression. The variables change in wetland area, change in glacier area, and change in lake area were derived from classified Landsat TM images acquired on 15 May 1987 and 6 June 1995 (downloaded from http://glovis.usgs.gov). For each date, two Level 1T images (path-row 8-66, 8-67) were mosaicked together. These dates were selected because they are relatively cloud-free and occur in the dry season when vegetation phenological differences are minimal. Level 1T is a processing level that provides systematic, radiometric, and geometric accuracy by using a digital elevation model for topographic accuracy. Image-to-image registration is consistent across the dataset and is less than one-half of one pixel (Hansen and Loveland 2012). Images were analyzed independently post-classification, thus atmospheric correction was unnecessary (Song et al. 2001).

Table 4.2. Variables and their descriptions.

Variable	Symbol	Definition and Ecological Meaning	Unit
DEPENDENT			
Wetland Area	WL_area	Area in catchment covered by wetland	ha
INDEPENDENT			
Glacier Area	Glacier_area	Area in watershed covered by snow/ice	ha
Lake Area	Lake_area	Area in watershed covered by lake surface	ha
Mean Annual Discharge	MeanAnnQ	Mean annual discharge from the calendar year coincident with image acquisition year	m^3/s
6-year Mean Annual Discharge	6MeanAnnQ	Mean annual discharge from the calendar year 6 years prior to image acquisition year (1981, 1989)	m^3/s
Dry Season Discharge	DryQ	The total discharge from the dry season preceding image acquisition dates (July and August 1986, 1994)	m^3/s
Wet Season Discharge	WetQ	The total discharge from the wet season preceding image acquisition dates (February and March 1987, 1995)	m^3/s
Total Annual Precipitation	PrePrior	The total precipitation computed from the 12 months preceding the image acquisition dates	mm
6-year Total Annual Precipitation	6prePrior	The total precipitation computed from the 12 months occurring 6 years preceding image acquisition dates (1981, 1989)	mm
Dry Season Precipitation	preJulyAug	Total precipitation from the dry season preceding image acquisition dates (July and August 1986, 1994)	mm
Wet Season Precipitation	preFebMar	Total precipitation from the wet season preceding image acquisition dates (February and March 1986, 1994)	mm

Table 4.3. Pooled summary statistics for all variables in 1987 and 1995. * represents missing data for La Balsa in 1981. See text for explanation.

Variable	Symbol	Mean	Standard Deviation	Minimum	Maximum
Wetland Area	WL_area	717.13	1303.65	39.96	3745.26
Glacier Area	Glacier_area	7983.51	13,641.12	238.32	43,834.86
Lake Area	Lake_area	335.46	494.39	66.06	1629.72
Mean Annual Discharge	MeanAnnQ	15.17	28.42	1.52	86.31
6-year Mean Annual Discharge	6MeanAnnQ	10.08	22.90	*	89.16
Dry Season Discharge	DryQ	6.00	9.72	0.29	32.06
Wet Season Discharge	WetQ	26.21	50.40	2.89	155.48
Total Annual Precipitation	PrePrior	373.91	49.94	284.90	464.10
6-year Total Annual Precipitation	6PrePrior	718.35	183.12	520.40	990.60
Dry Season Precipitation	PreJulyAug	12.27	21.03	0.50	82.50
Wet Season Precipitation	PreFebMar	142.29	54.28	72.90	222.40

Similar to Chapter 3, a hybrid supervised-unsupervised image classification was performed (using ERDAS Imagine 2014), a technique that uses all 7 bands in 30 m spatial resolution (Messina, Crews-Meyer, and Walsh 2000; Walsh et al. 2003). The classification technique was selected for its reliable previous performances to measure land change in the Andes (Kintz, Young, and Crews-Meyer 2006; Lipton 2008; Postigo, Young, and Crews 2008). First the unsupervised classification method (ISODATA) clusters the spectral information into 255 signatures. These signatures are then evaluated for separability using the transform divergence method, removing signatures with low spectral separability. Next, the edited signatures are applied in the supervised classification. To improve the accuracy of difficult-to-classify wetlands, I added a 4-3 band ratio (Ozesmi and Bauer 2002). Lake colors vary widely due to varying amounts of suspended glacial flour resulting in poor classification by the hybrid method. Therefore, lakes were digitized manually for each time period using the GLIMS dataset (http://www.glims.org/) as a basis. Known for its glacier inventory, the GLIMS dataset also includes lake features (Kargel et al. 2014). The final product was a thematic map for each date with 7 land cover types: Barren, Puna, Wetland, Snow/Ice, Lake, Shadow, and Cloud (refer to Chapter 3 for detailed descriptions). For the purposes of this research, only Wetland, Snow/Ice, and Lake were used. Accuracy assessments were within standard acceptability norms over 85% (Khorram 1999; Congalton and Green 2008). For 1987, overall accuracy was 89.38% and overall Kappa was 87.52%. For 1995, overall accuracy was 85.4% and overall Kappa was 82.3% (See Appendix 4 for Producers and Users Accuracies). Ideally, accuracy assessments would be performed on all images, but reference data that meet accepted criteria (higher spatial resolution, acquired at approximately the same time of year, same spectral resolution) was unavailable for the 2 images. As an alternative, the images themselves were used as the reference data. Classification errors - although small - replicate through the analysis and are recognized as possible sources of error.

Precipitation and temperature variables were derived from the Climatic Research Unit dataset, CRU TS 3.10.01 (University of East Anglia Climatic Research Unit, Jones, and Harris 2013) and downloaded from CGIAR-CSI (http://www.cgiar-csi.org). CRU data offer deep temporal resolution from 1901 to 2010 at 0.5° lat/long grid cells. The CRU climate variables were synthesized from monthly observations from meteorological stations across the global terrestrial surface, excluding Antarctica (Mitchell and Jones 2005; Harris et al. 2014). Others have used CRU data in the Cordillera Blanca (Vuille, Kaser, and Juen 2008; Urrutia and Vuille 2009) in spite of acknowledgements that the data should be used with caution because there are no inputs from weather stations in Peru and it may overestimate temperature. For 1987 and 1995, four precipitation variables were created that were meant to account for the lagged effect of surface flow and groundwater moving into peatlands (Table 4.2). For the 12 months preceding the image acquisition date, total annual precipitation was calculated. A second 6-year lagged precipitation variable was created for the 12 months prior to the image acquisition date (1981 and 1989). Wet season precipitation is the sum of the two months with historically highest precipitation, February and March, preceding the image acquisition dates (Baraer et al. 2012). Dry season precipitation was calculated from the sum of the two months preceding the image acquisition dates with the historically lowest precipitation, July and August. The temperature variable is the mean for the 12 months preceding the image acquisition date; due to inadequate variation across the two time periods, temperature was omitted from the analysis. Where watersheds contained more than one CRU grid cell, the mean value was used. Ideally, in situ climate and hydrologic data would be collected within each of the 7 catchments to account for local variation characteristic of mountain geography, but the Peruvian Andes are poorly gauged. Consequently, meteorological and hydrologic datasets suffer from periodic interruptions and incomplete records make temporal analysis challenging. As such, the CRU time series represents the best available climate data in terms of spatial coverage, temporal depth, and consistency. The Tropical Rainfall Measuring Mission dataset was not used because the temporal record begins in 2000, 5 years after the discharge record used for this study.

Discharge data (Q) originated from a collection of records dating to the early 1950s, when an array of stream and precipitation gauges was installed in the Santa River basin. The historical dataset originally included daily time series for 17 stations and after quality control analysis, only 9 stations were determined to be suitable for trend analysis (refer to Baraer et al. 2012 for screening criteria and procedures): Chancos, Colcas, La Balsa, La Recreta, Llanganuco, Los Cedros, Pachacoto, Parón, and Querococha. Of these 9 stations, 2 were eliminated for the present study because 1) Parón contains a glacial lake that is regulated by a drainage tunnel with flows intensively managed by a hydroelectric power generation firm so discharge is artificially biased; and 2) Pachacoto was eliminated because the discharge record was interrupted from 1992-1993 and in 1995. Four stream discharge variables were calculated (Table 4.2). Mean annual discharge and dry season discharge were selected because they have been shown to be markers of the influence of glacier retreat on the hydrology of medium to large watersheds in the study area (Baraer et al. 2012). Mean annual discharge is the average discharge for the calendar year coinciding with the image date. A 6-year lagged discharge variable includes discharge from 1981 and 1989 to account for residence times (Baraer et al. 2014). Dry season discharge is defined as the total discharge for the 2 months with the historically lowest precipitation (July and August) coinciding with the image acquisition year. Wet season discharge is the total discharge for the 2 months with the historically highest precipitation (February and March) preceding the image acquisition date. A geographic information system (ArcGIS 10.2) was used with an ASTER GDEM v.2 digital elevation model (http://gdem.ersdac.jspacesystems.or.jp/) to generate watersheds based on the coordinates of the 7 discharge stations. As an example, Figure 4.3 illustrates land change in the Querococha watershed from 1987 to 1995.

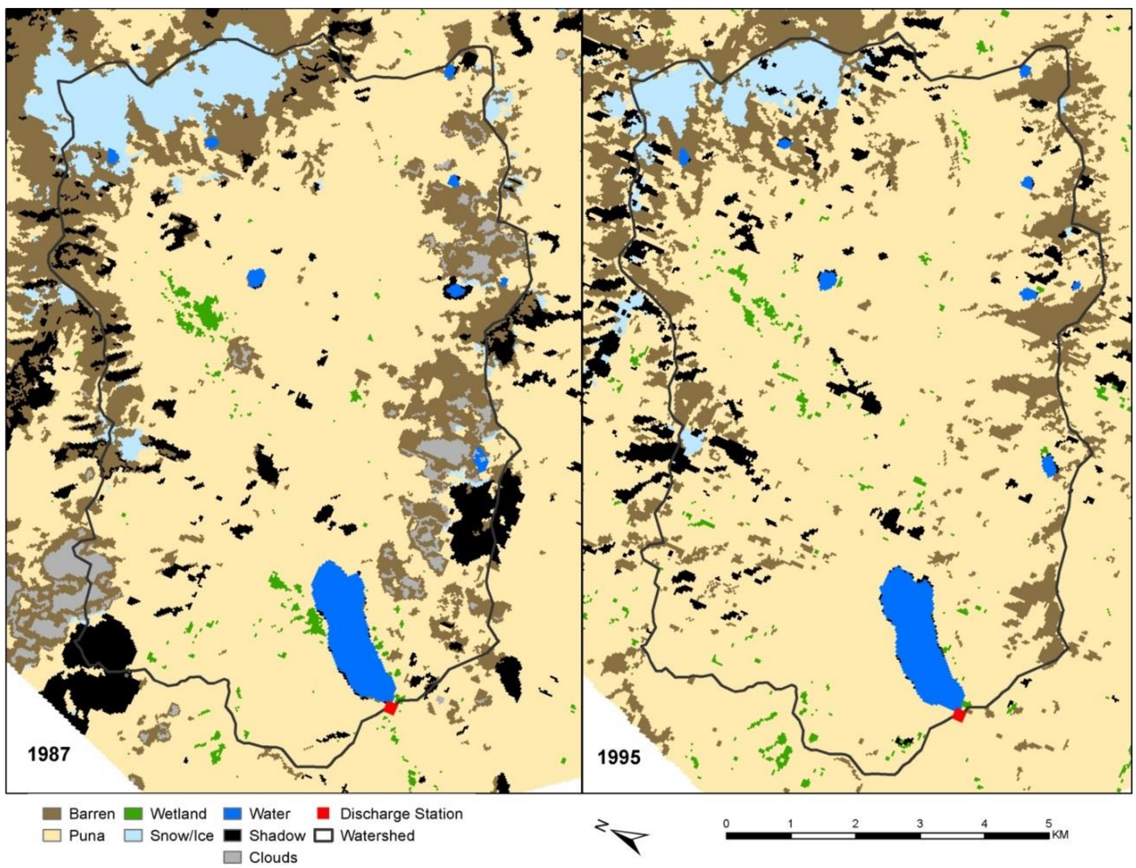


Figure 4.3. The Querococha watershed in 1987 (L) and in 1995 (R).

For a panel dataset with observation *i* and 2 time periods (t = 1 or 2), the FD regression equation is expressed as:

$$y_{i2} = (\beta_0 + \delta_0) + \beta_1 x_{i2} + a_i + u_{i2} \text{ (time = 2)}$$

$$y_{i1} = \beta_0 + \beta_1 x_{i1} + a_i + u_{i1} \text{ (time = 1)},$$

where y_{it} is the dependent variable of interest, β_0 is the y intercept or constant, $\beta_0 + \delta_0$ is the change in the intercept from time 1 to time 2, β_1 is a coefficient of predictor variable x, x_{it} represents predictor variables, a_i is the unobserved time constant factor (the unobserved effect), and u_{it} is the time-varying error (Wooldridge 2009). After subtracting time 1 from time 2, the following equation results:

$$(y_{i2} - y_{i1}) = \delta_0 + \beta_1(x_{i2} - x_{i1}) + (u_{i2} - u_{i1})$$

or

$$\Delta y_i = \delta_0 + \beta_1 \Delta x_i + \Delta u_{i2}$$

where Δ represents the change from time 1 to time 2. The term "first differencing" was coined because the unobserved effect, a_i is "differenced away" so that the time-constant unobserved effects are controlled for (Wooldridge 2009, 458). The rationale is that time-constant variables are not likely to be factors driving change (Allison 2009; Andreß, Golsch, and Schmidt 2012). Statistical computations were conducted in STATA SE 12 with the function xtreg and the fe and vce(robust) options for fixed effects and robust standard errors. The following four main assumptions for fixed effects regression have not been violated, implying that model estimates are efficient and unbiased (Andreß, Golsch, and Schmidt 2012):

1) Exogeneity – the unobserved variables and error term are independent of one another.

2) Homoskedasticity – both the unobserved variables and the error term have constant variance. Robust standard errors were computed to control for possible presence of heteroskedasticity.
3) No serial correlation – the idiosyncratic errors are independent from one another.
4) No measurement error – the observed values are identical to the true values of the dependent variable, independent variables, and errors.

Model specifications were performed in two stages. First, FD bivariate regressions of change in wetland area on all the independent variables were computed. Second, FD regressions were completed following the 7 hypotheses (Table 4.1, Figure 4.1). The significance value selected was 0.1 (two-tailed). Although the significance value is higher than the often-used 0.05 level, I selected 0.1 for two reasons. First, the small number of observations (n = 7) necessitates a looser criteria. Second, applying FD to tropical high altitude wetlands is an innovative effort that is intended to explore and potentially advance the efficacy of econometrics in land change science. In the spirit of transparency, actual significance levels are included for the reader's reference in Tables 5.4, 5.5, 5.6, and 5.7. Using the command xtreg, STATA produces three R^2 values: within, between, and overall. The within R^2 is "usual R^2" (Allison 2009, 19) and the most commonly used for interpreting FD regressions. It reports the explained portion of the within variance using the mean deviation variables (Allison 2009; Andreß, Golsch, and Schmidt 2012). All R^2 values reported herein are the within R^2s.

RESULTS

After computing the bivariate regressions and 7 models, I found that prior precipitation was statistically significant, but that change in glacier area and mean annual discharge are better predictors of wetland change. First, the bivariate regressions were computed to test H1 and to determine whether or not any of the other independent variables might individually explain change in wetland area (Table 4.4). One variable, prior precipitation (PrePrior) was highly statistically significant ($p < 0.001$, two-tailed) and the within R^2 was 0.5571. In spite of the high statistical significance of PrePrior, it is unlikely that a single variable adequately explains change in wetland area. Simple regressions usually suffer from omitted variables (Wooldridge 2009). As such, it is highly unlikely that prior precipitation can alone predict change in wetland area.

Table 4.4. Bivariate regressions of change in wetland area on independent variables. Robust standard errors are shown.

Variable	**Constant**	**Robust SE**	**Coefficient**	**Robust SE**	**p value**	**R^2 Within**
Δ Glacier Area	712.4923	17.4413	0.0005	0.0022	0.799	0.0009
Δ Lake Area	728.9068	28.7390	-0.0351	0.0856	0.696	0.0035
Δ Mean Annual Q	684.5593	58.5739	2.1468	3.8604	0.598	0.0137
Δ 6 Yr Mean Annual Q*	223.8707	130.0444	-2.5637	30.0275	0.935	0.0035
Δ Dry season Q	719.7329	21.2511	-0.4332	3.5414	0.907	0.0004
Δ Wet season Q	696.8809	31.6755	0.7726	1.2085	0.546	0.0108
Δ PrePrior	-437.3186	157.0036	3.0874	0.4199	0.000	0.5571
Δ 6 Yr PrePrior	688.5121	61.4422	0.0398	0.0855	0.658	0.0421
Δ Pre July Aug	712.1690	5.3583	0.4042	0.4363	0.390	0.0278
Δ Pre Feb Mar	701.4043	42.1073	0.1105	0.2959	0.722	0.0280

* Does not include La Balsa watershed; n=6

The results of the multivariate FD regressions are reported in Table 4.5. None of the models pass the stated acceptance criteria ($p < 0.1$) with the exception of Model 3; therefore Hypotheses (and Models) 1, 2 and 4-7 are rejected. Model 3 fails to be rejected and shows that change in glacier area and change in mean annual discharge were statistically significant at the 0.1 level. Within R^2 was 0.3431. Multicollinearity was detected among the following variables: mean annual discharge, dry season discharge, and wet season discharge. Similarly, multicollinearity existed among previous precipitation and dry and wet season precipitation. To eliminate effects of multicollinearity, models were restricted to mean annual discharge and previous precipitation.

Table 4.5. Model results for multiple regressions. Model numbers correspond with hypotheses described in Table 4.1 and Figure 4.1. Robust standard errors reported in parentheses below coefficient.

Variable		**Model 1**	**Model 2**	**Model 3**	**Model 4**	**Model 5**	**Model 6**	**Model 7**
Constant	Coeff	712.4923	2327.0300	218.9284	1262.1010	-566.9294	-1279.7660	-504.7602
	S.E.	(17.4413)	(2397.4540)	(219.3361)	(2139.7250)	(411.7595)	(1823.6730)	(220.8903)
	p value	0.000	0.369	0.357	0.577	0.218	0.509	0.062
Δ Glacier Area	Coeff	0.0005	-0.0874	-0.0743	-0.1207	.01851	0.0582	
	S.E.	(0.0022)	(0.1303)	(0.0303)	(0.0694)	(0.0860)	(0.1006)	
	p value		0.527	0.050	0.133	0.837	0.584	
Δ Lake Area	Coeff		-2.7187		-1.6717		0.9693	
	S.E.		(4.0455)		(3.1715)		(2.5702)	
	p value		0.527		0.617		0.719	
Δ Mean Annual Q	Coeff			71.9332	64.5858	-22.1651	-30.8301	-3.8722
	S.E.			(30.4054)	(40.2777)	(86.5298)	(81.0252)	(2.9604)
	p value			0.056	0.160	0.806	0.717	0.239
Δ Pre Prior	Coeff					3.9381	4.4791	3.4249
	S.E.					(2.6804)	(2.6185)	(0.6652)
	p value					0.192	0.138	0.002
R^2 Within		0.0009	0.1334	0.3431	0.3896	0.6159	0.6264	0.6088

Furthermore I wanted to explore a longer lagged discharge variable in the event that the 12 month lagged discharge variable, Mean Annual Discharge (MeanAnnQ, Table 4.2), did not capture the interaction. In other words, is the hydrologic residence time longer than the previous 12 months? To address this question, a 6-year lagged discharge variable was created. Six years was selected because the only preceding years in which the discharge data were gap free were 1981 and 1989. La Balsa was omitted because of missing data in 1981 so in this case n = 6. The 6-year lagged variable was substituted into Model 3 and is referred to as Model 3A. The analysis revealed that the 6-year lagged discharge variable did not improve the estimations (Table 4.6). Six year mean annual discharge and change in glacier area were not statistically significant and the R^2 value declined to 0.0377.

Table 4.6. Results for FD regressions on Model 3 and 3A. Model 3A excluded La Balsa (n = 6) and substituted a 6-year lagged discharge variable for 12-month discharge variable. Model numbers correspond with hypotheses described in Table 4.1 and Figure 4.1. Robust standard errors reported in parentheses below coefficient.

Variable		**Model 3**	**Model 3A**
Constant	Coeff	188.7638	175.0493
	S.E.	(143.7029)	(183.5832)
	p value	0.246	0.384
Δ Glacier Area	Coeff	-0.1229	0.0399
	S.E.	(0.0743)	(0.1068)
	p value	0.159	0.723
Δ Mean Annual Q	Coeff	88.5584	
	S.E.	(34.8529)	
	p value	0.052	
Δ Mean Annual Q 6	Coeff		-16.4203
	S.E.		(46.1591)
	p value		0.737
R^2 Within		0.4110	0.0377

Finally, I examined the effect of omitting the La Balsa watershed because it is substantially larger than all 6 other watersheds (479,000 ha versus the next largest watershed, La Recreta at 28,000 ha). After removing the observation (n = 6), I tested Model 3 because it is the best estimate of the 7 hypotheses. With fewer observations, the within R^2 value increased from 0.3431 to 0.4110 (Table 4.7). In Model 3A, only change in mean annual discharge is statistically significant (p = 0.052). Comparing the independent variable coefficient estimates with and without La Balsa, the magnitudes and signs were consistent. Omitting La Balsa does not appear to have a major effect on the analysis. Given the loss of degrees of freedom by omitting La Balsa, the watershed should be included to boost confidence in the model's predictive abilities.

Table 4.7. Results for FD regressions including (Model 3) and excluding La Balsa (Model 3A, n=6) and substituting a 6-year lagged discharge variable for 12-month discharge variable. Model numbers correspond with hypotheses described in Table 4.1 and Figure 4.1. Robust standard errors reported in parentheses below coefficient.

Variable		**Model 3 With La Balsa**	**Model 3A Without La Balsa**
Constant	Coeff	218.9284	188.7638
	S.E.	(219.3361)	(143.7029)
	p value	0.357	0.246
Δ Glacier Area	Coeff	-0.0743	-0.1229
	S.E.	(0.0303)	(0.0743)
	p value	0.050	0.159
Δ Mean Annual Q	Coeff	71.9332	88.5584
	S.E.	(30.4054)	(34.8529)
	p value	0.056	0.052
R^2 Within		0.3431	0.4110

DISCUSSION

The first differencing regression used in this analysis indicated that change in glacier area and change in mean annual discharge were driving wetland change from 1987 to 1995. Results from Model 3 estimated that for every 1 ha decrease in glacier area, wetland area increased by 0.07 hectares, holding all other factors constant. Relative to change in glacier area, change in mean annual discharge accounted for the largest magnitude of wetland change from 1987 to 1995. For every 1 m^3/sec increase in annual discharge, wetland area increased by 72 ha, holding all other factors constant. The constant coefficient in Model 3 was not statistically significantly different from zero, so in the absence of melting glaciers and changing discharge, expected wetland change in the study area would be minimal. In addition, there is a possible role of increased precipitation increasing wetlands, perhaps by augmenting area in wet meadows and peatlands.

The indications that changing glacier area and changes in mean annual discharge were associated with change in wetland area need to be evaluated within the context of the “peak water” concept published by Baraer et al. (2012) and first proposed by Jansson et al (2003). Using the same discharge data used in this study, they proposed that discharge from de-glaciating watersheds in the Cordillera Blanca followed a 4-phase trajectory. In the first phase, discharge increased concurrently with accelerated glacier recession to levels above previous historic flows. In phase 2, discharge increased but reached a peak flow, or “peak water” (Bury et al. 2013, 367). Phase 3 is characterized by a distinct decrease from the peak in phase 2. Finally in phase 4, discharge reached a new lower equilibrium; meanwhile, discharge variability increased. Of the nine watersheds evaluated by Baraer et al. (2012), seven were in phase 3, one in phase 4 and one in phase 1 at the present time. The seven watersheds used in this study were all in phase 3, with the exception of La Recreta in phase 4. Six of the seven watersheds had entered phase 3 by 1987 and one entered phase 3 in 1988.

In other words, the watersheds in this study were already past “peak water” and discharge was beginning to decline by 1987-1988. Model 3 estimated a positive relationship between change in mean annual discharge and change in wetland area from 1987 to 1995. All seven watersheds were experiencing decreasing discharge and higher variability and wetlands responded by decreasing in spatial extent. With both variables moving in the same direction, mean annual discharge decreased which drove decreases in wetland area, controlling for changes in glacier area. The effects of reduced discharge and higher variability should be observable on the spatial distribution and configuration of wetland patches. With decreased hydrological input into wetlands in the future, soil desiccation is a likely scenario, as shown in Chapter 3. Observable spatial effects could include wetland fragmentation, attrition, dissection, shrinkage, and perforation. These patch modifications would likely result in associated ecological effects including loss of plants and animals adapted to or dependent on saturated substrates. Ecosystem services provided by the wetlands would decrease (Figure 4.4).

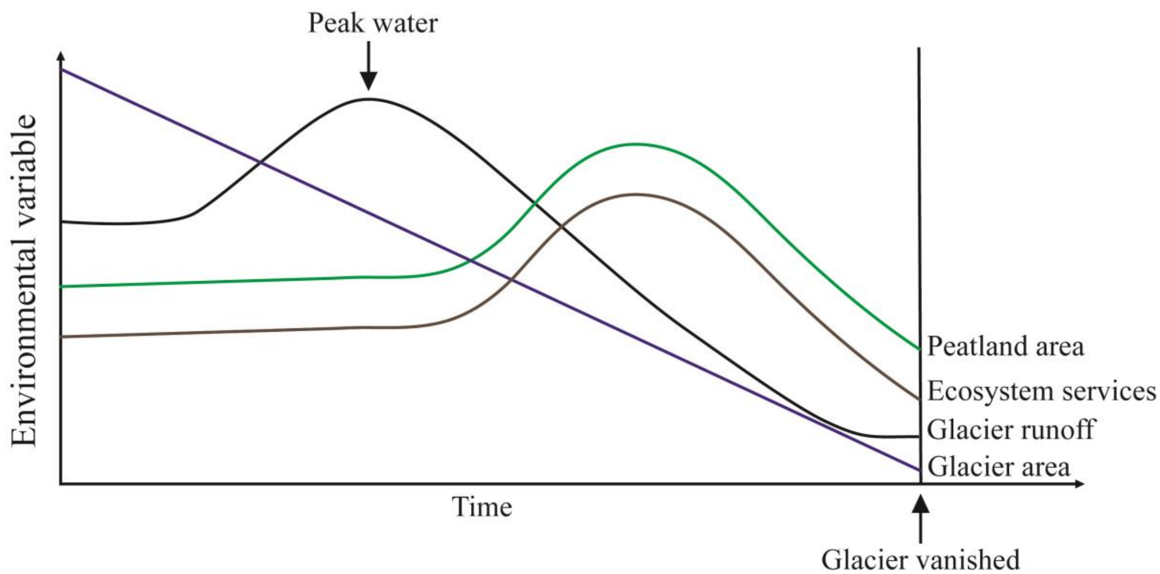


Figure 4.4. Conceptual graphical relationship between glacier area, glacier runoff peatland area, and ecosystem services.

The estimates reported here hold true for 1987 to 1995, but would likely differ if a nonlinear relationship between mean annual discharge and change in wetland area is generally to be expected over a longer period. For example, researchers working in the same area (Mark and Seltzer 2003; Baraer et al. 2012) have projected that precipitation will become increasingly dominant in the hydrology of the Cordillera Blanca watersheds because the effect of glacier meltwater will diminish over time. The relationship between changes in wetland area and prior 12-month precipitation supports this prediction (p-value is highly statistically significant; within R^2 is 0.5571, Table 3). Since wetlands in the study area are a complex of bogs, fens, and minerogenous meadows, it is conceivable that a future scenario might involve transitioning to increased dominance of precipitation-dependent bogs. Given the results in Chapter 3, precipitation based bogs will most likely be smaller and subject to processes observed over the last 23 years including fragmentation, attrition, isolation, and shrinkage.

CONCLUSION

Dynamic landscapes in the high elevation tropics are shaped by human activity, steep environmental gradients, and associated feedbacks and interactions among these processes. In watersheds where tropical glaciers are receding, new hydrologic regimes are emerging, causing ecological and social shifts. Connectivity between land change and tropical hydrologic processes are not well understood (Wohl et al. 2012; Ponette-González et al. 2014), but this study shows that a spatially explicit econometric analysis demonstrates a promising methodology that can be further developed for use by land change scientists.

In the context of changing landscapes and global environmental change in the high elevation tropics, the analysis provides insight into the hydrologic connectivity of wetlands in deglaciating watersheds by determining the relative effect of 2 factors on wetland change. These conclusions do not imply that glacier recession is unimportant; nor do they minimize the impact glacier recession will have on local populations and livelihoods. On the contrary, glacier recession is an enormous problem and the consequences are heterogeneous, impacting both biophysical and social systems. What this research reveals is a new understanding of land change and hydrologic connectivity in tropical glacierized watersheds. It further emphasizes the necessity of installing a network of meteorological instrumentation, stream gauges, as well as maintaining and upgrading our constellation of earth-observing satellites. Doing so would allow future researchers to employ FE regression on more than 2 time periods and use a larger sample size than was available for this study. With deeper temporal resolution and larger sample sizes, establishing causality with confidence may be attainable.

The capacity to establish causality and make prediction is especially important in tropical landscapes that face a triple threat from global environmental change, population growth, and intensified natural resource extraction. As land managers and policy makers prepare for change associated with glacier recession, statistical methods such as FE regression offer a tool with predictive capacities. Because these wetlands store high volumes of carbon, harbor biodiversity, and as shown here, are a critical component in the hydrologic connectivity of tropical watersheds, they deserve to be protected against the suite of threats. Understanding the spatial driving forces of change would allow managers to better protect and restore these ecosystems.

Chapter 5: Vegetation Ecology of Tropical Mountain Peatlands

INTRODUCTION

Tropical high altitude peatlands are found in the mountains of East Africa, New Guinea, Hawaii, and across the Andes spanning from Venezuela to Chile. Recent publications have highlighted their importance in the Andes where they are known as *bofedales*, *oconales*, or *humedales* (Cooper et al. 2010, 2015; Buytaert, Cuesta-Camacho, and Tobón 2011; Benavides 2014; Salvador, Monerris, and Rochefort 2014; Izquierdo, Foguet, and Grau 2015; Maldonado Fonkén 2015). For example, they are key components in montane hydrology, storing precipitation and groundwater in highly porous soil (Harden 2006; Gordon et al. 2015) and regulating the release of stored water seasonally (Buytaert et al. 2006) thus buffering against low stream discharge during the dry season. Biologically, they have high plant endemism and diversity, and plants and wildlife are specifically adapted to live in conditions unique to high altitude peatlands (Young, Young, and Josse 2011; Zinck 2011). Ecologically, peatlands function as carbon pools, are sources of methane, produce biomass, and transport and filter nutrients (Charman 2002; Chimner and Karberg 2008; Cooper et al. 2015). In this context of high hydrological, biological, and ecological significance, this chapter explores the vegetation ecology of tropical mountain peatlands in Peru's Huascarán National Park (HNP).

Peru is regarded as a biodiversity hotspot (Myers et al. 2000) and the tropical Andes are home to exceptionally high species diversity (Luteyn and Churchill 2000). New species are routinely discovered at altitudes > 4000 m.a.s.l. in the Peruvian Andes, including zones in and around HNP (Cano et al. 2010; Al-Shehbaz, Cano, and Trinidad

2012; Al-Shehbaz, Navarro, and Cano 2012). To date, the most comprehensive taxonomic study of HNP was completed by David N. Smith (1988) and published as a doctoral dissertation. Smith's work was a continuation of a 1984 Missouri Botanical Garden project to catalog the park's flora, representative of the jalca, an ecosystem located between the wetter páramos to the north and the drier puna to the south (Weberbauer 1945; Young et al. 1997). Among his research justifications, Smith (1988) noted that previous taxonomic work had occurred north of the equatorial Andes and HNP was representative of a variety of high mountain vegetation in Peru that could be compared with floras of the western and eastern Cordilleras. In addition to Smith's dissertation, one other field guide for the east side of the park covers part of the flora of HNP (Cano et al. 2006). What these works do not specifically address is the flora of high mountain peatlands, which is one of the goals of this chapter. Furthermore, of the species that are present in HNP's high mountain peatlands, it is not known which species most often occur together. Documenting these plant communities is important because peatlands are changing spatially and are thus presumed to be impacted ecologically by fragmentation, attrition, shrinkage, and isolation (See Chapter 3 and Bury et al. 2013). Qualitatively, Smith (1988) reported that cushion plants in HNP are associated with species in the Cyperaceae, Asteraceae, and Poaceae families. Building on Weberbauer's work (1945), Maldonado Fonkén (2015, 4) described plant species composition within four hydrophitic plant communities typical for all Peruvian bofedales: *Distichia* peatland, peatland with mosses and shrubs, peaty meadow, and stream grassland. Documenting whether or not these same communities – or others to be described – exist in the study area is another goal of this chapter.

Understanding what controls the organization of peatland vegetation is an area of ongoing research. Broadly speaking, peatland vegetation composition is thought to be controlled by two complex environmental gradients: a) variation in wetness and soil aeration, and b) variation in pH, base richness and nutrient availability (Rydin and Jeglum 2013). Spatial and temporal fluctuations in the water table create gradients of soil wetness. As pore spaces fill with water and drain over time, plants must respond physiologically to these water table variations. Oxygen levels in the rooting environment influence both biomass production and decomposition rates, which has direct effect on vegetation type and distribution. As peat accumulates through decomposition, organic matter (OM) increases, and pH and nutrient levels decrease. Peat cation exchange capacity (CEC) is typically high because of an abundance of hydrophilic colloids, especially humic acids and hemicelluloses (Zinck 2011; Rydin and Jeglum 2013). Peat particles adsorb most cations leaving few available for uptake by plants roots. Although nitrogen is typically the primary limiting nutrient in most terrestrial environments, phosphorus (P) and potassium (K) are the limiting nutrients in peatlands. Low P is believed to be a reason why trees and shrub growth are restricted in peatlands (Rydin and Jeglum 2013), although fertility and pH levels also play a role in determining peatland vegetation types (Keddy 2010).

A review of the literature on factors controlling vegetation in Andean peatlands illustrates that there is no general consensus on which factors predominate. Instead, complex combinations of gradients are at work and vary across the Andes. In the Colombian páramo, Bosman and others (1993) found that peat thickness, N and iron concentrations, electric conductivity, and pH controlled vegetation composition. Another study from the Colombian páramo pointed to elevation, precipitation, and pH gradients (Benavides and Vitt 2014). In northern Chile, Squeo and others (2006) propose that vegetation is controlled by several interacting factors: water quantity and seasonal availability, temperature and frost occurrence, pH, availability of nutrients (N, P, K, Ca, Mg), and exposure to toxic elements (As, B, Fe, Al). Working in Torres del Paine National Park in southern Chile, Clausen and others (2006) found that dissolved minerals, redox conditions, and precipitation are controlling factors. Biotic variables such as grazing, animal seed dispersal, and human impacts (drainage channels, agricultural activity) also played a role in shaping vegetation composition (Squeo et al. 2006; Tovar et al. 2012; Benavides 2014). Two studies from the Peruvian jalca found different factors controlling vegetation. Cooper and others (2010) reported that floristic composition is driven by groundwater chemistry (HCO_3 and pH) and a complex gradient of hydrologic, soil temperature, and peat thickness. Tovar and others (2012) identified pH, elevation and peatland patch size to be the controlling factors. Given these differing results and the documented spatial changes, understanding the controlling factors of peatland vegetation in HNP is justified. Based on the work by Tovar and others (2012) and Cooper and others (2010) in similar and nearby peatlands, elevation and pH was hypothesized to explain vegetation composition. High water acidity has been documented in the study area: in the Quilcayhuanca watershed, more than 70% of stream samples were pH < 4. Receding glaciers have exposed sulfide-rich rocks that change stream biochemistry (Fortner et al. 2011) that could then impact chemical characteristics of peatlands.

The results presented in this chapter build on and enhance existing research by focusing on peatland vegetation ecology in HNP. The three goals of this chapter are as follows: 1) Compile a species list for HNP peatlands, evaluate richness and diversity, and determine inter-valley similarity and dissimilarity; 2) Characterize plant communities occurring in peatlands; 3) Identify the abiotic factors that control vegetation composition. Overall the research fills a geographic and ecological knowledge gap in floristic diversity in the Peruvian puna that is affected by climate change and glacier recession.

METHODS

The study was conducted in 3 valleys inside the boundaries of Huascarán National Park: Llanganuco, Quilcayhuanca, and Carhuascancha (Figure 5.1). Field work was conducted in three episodes during the dry season (June-July) in 2012, 2013, and 2014 by the author, Kenneth R. Young, Blanca León, and Asunción Cano Echeverría under SERNANP Permit TUPA N° PNH-008-2012 valid from 1 July 2012 to 30 June 2015. All vegetation sampling was completed in 2012 and 2013; soil sampling was completed in 2013 and 2014. Changes from year-to-year should be minimal and are not expected to complicate analyses. A total of 65 2 x 2 m quadrats were stratified by elevation. To minimize bias, quadrats were located by a blind reverse overhead toss into a relatively homogeneous peatland vegetation patch. Species data for vascular plant taxa were collected in every quadrat and environmental variables were collected in every other quadrat. Species data included botanical name, percent cover using the Domin scale

(Kent 2012), and height and name of tallest species. Unknown specimens were collected and later verified against vouchers at the Museo de Historia Natural, Universidad Nacional Mayor de San Marcos in Lima, Peru by a recognized taxonomic authority, Asuncíon Cano Echeverría. The following environmental variables were collected: GPS coordinate (UTM Zone 18S, WGS 84, Garmin GPSmap 62), elevation, slope, aspect, peat depth, and wetness index. The wetness index was adapted from Lemly and Cooper (2011) and ranged from 1 to 4 where 1 = saturated soil, hummocks, and no standing water; 2 = saturated soil, no standing water; 3 = < 25 cm standing water; 4 = dry soil. Depth and soil samples were collected on every second quadrat (n = 34). Depth was measured with a 3/8" steel tile probe that was pushed into the peat soil until rock or an impenetrable substrate was encountered. A 1 kilogram soil sample was collected at 10 cm depth and later analyzed at the Laboratorio de Suelos at the Universidad Nacional Agraria La Molina in Lima, Peru. The following chemical and physical factors were analyzed: pH, percent organic matter, phosphorus, potassium, textural class, cation exchange capacity, and bulk density. Methods employed by the Laboratorio de Suelos for these factors are detailed in Table 5.1. Water chemistry is recommended for peat soils (Rydin and Jeglum 2013), but in this case soil chemistry was analyzed because it is a better correlate for vascular plant composition (Sjörs and Gunnarsson 2002). The full dataset was recorded in 2 matrices: the species matrix containing 112 species x 65 plots; and the environmental matrix containing 13 variables x 65 plots. Summary statistics for the environmental variables are found in Table 5.2 and full results of the soil analysis are found in Appendix 5.

Figure 5.1. Images of the three valleys: Llanganuco (L), Quilcayhuanca (C), and Carhuascancha (R)

Table 5.1. Methods used by the Laboratorio de Suelos, Universidad Nacional Agraria La Molina for 1 kg soil samples.

Factor	Method
pH	Potentiometer of soil suspension to water ration in 1:1 relation
% Organic Matter	Oxidation of organic carbon with potassium dichromate (Walkley and Black)
Phosphorus	Extraction of with $NaHCO_3$ 0.5N, pH 8.5 (modified Olsen method)
Potassium	Extraction with ammonium acetate, spectrophotometry of atomic absorption
Textural class	Quantification of sand, loam, clay using hydrometer
Cation Exchange Capacity	Saturation of the clay-humic complex with ammonium acetate and posterior distillation with nitrogen (Kjeldahl)
Bulk Density	Soil clod in paraffin

Table 5.2. Summary statistics for environmental variables.

	Min	Max	Mean	Median	Std Dev
Elevation (m.a.s.l.)	3827	4629	4148	4166	221.7
Slope (°)	0	26	5	4	4.2
Peat depth (cm)	0	8320	321	95	1230.2
pH	3.6	5.4	4.4	4.5	0.4
Organic Matter (%)	1.8	81.5	46.3	55.3	24.1
P (ppm)	0.1	260	41.6	13.8	56.9
K (ppm)	26.0	1053.0	348.8	289.0	280.6
CEC (meq/100g)	3.2	80.0	45.5	51.2	24.4
Bulk Density (g/cm^3)	0.1	1.2	0.3	0.1	0.4

Quantitative data analyses were performed in PC-ORD v. 6.19 (McCune and Grace 2002) except for the Steinhaus (Sørenson/Czekanowski) coefficient, calculated in Microsoft Excel. Kent (2012) reports the Steinhaus coefficient of similarity as:

$$Ss = \frac{2a}{2a + b + c}$$

Dissimilarity is given as:

$$Ds = 1.0 - Ss$$

where a is the number of species common to both valleys, b is the number of species in valley 1 only, and c is the number of species in valley 2 only (Kent 2012). To test the hypothesis of no difference among valleys, Multi-Response Permutation Procedures

(MRPP in PC-ORD v. 6.19) were applied using Sørenson (Bray-Curtis) distance and groups defined by valleys. Correlation between richness and elevation and height and elevation was computed using Spearman's rank correlation coefficient that measures the strength and association between two continuous variables. It is used when the relationship between the two variables is non-linear and when normality cannot be assumed.

Peatland plant communities were separated by a hierarchical, polythetic, agglomerative cluster analysis (Cluster Analysis in PC-ORD v. 6.19, Euclidean distance and Ward's measure of linkage) using abundance data in a transpose matrix (species in rows, quadrats in columns). A strict criterion for species inclusion was used in the cluster analysis because species with few occurrences do not provide enough information to form groups (McCune and Mefford 2011). To be included, a species had to have occurred in 5 or more quadrats; consequently, 38 species were used in the cluster analysis. The data matrix contained 38 species x 65 plots. Using absolute abundance data tends to group species together by abundance; therefore abundance data for the cluster analysis were relativized by species sums of squares to remove the influence of absolute abundance (McCune and Mefford 2011).

To identify the abiotic factors controlling vegetation composition, Non-Metric Multidimensional Scaling was used (NMS in PC PC-ORD v. 6.19) with the Kruskal (1964) and Mather (1976) algorithms. NMS is an ordination technique that uses rank order information in a data matrix of dissimilarities among quadrats or species (Kent 2012); quadrats were used in this study. Based on the number of dimensionless axes specified by the analyst, NMS produces a monotonic distance matrix that is then displayed graphically in an ordination diagram. Quadrats displayed close to other quadrats in the ordination diagram have similar species compositions (McCune and Mefford 2011). The axes of the ordination diagram represent close associations with environmental variables. These associations are best interpreted visually by creating a joint bi-plot that overlays statistically significant correlations onto the ordination space thus revealing close associations between quadrats and environmental variables (Peck 2010). The environmental variables are represented by vectors that radiate out from the center of the diagram and point in the direction of influence. The length of the vector is positively related to the strength of its influence to the samples. NMS has become the most popular technique for community ecology studies because: a) it does not assume linear relationships between samples; b) it overcomes the zero-rich matrix problem typical of ecological data by using ranked distances; and c) it is inherently flexible because any distance measure can be used (McCune and Mefford 2011; Kent 2012)

In this case, where the goal is to determine the factors controlling vegetation composition, the ordination was performed on a subset of the complete dataset, 34 quadrats where soil samples were collected. The same strict criterion for species inclusion was employed (occurrence in > 5 quadrats). Thirteen abiotic variables were included. Continuous variables were elevation, slope, pH, % organic matter, phosphorus, potassium, cation exchange capacity, bulk density, peat depth, and species richness. Categorical variables were aspect, wetness index, valley number (1, 2, 3), and pH (binned into quartiles). The species matrix contained 34 plots x 38 species and the environmental matrix contained 34 plots x 16 variables. Transformation and relativization of the data was deemed unnecessary because there was a small range in percent cover across the 34 plots (42% - 97%) and the goal is to allow differences among plots to be expressed in the analysis, which would have been muted by relativization and transformation (McCune et al. 2000). The distance measure was Sørenson (Bray-Curtis), commonly used in vegetation studies when data are zero-rich and heterogeneous and outliers influence the results (McCune and Mefford 2011).

Three preliminary NMS runs using the Autopilot feature were completed to determine the optimum number of axes. I used a step-down approach starting with 6 axes, 500 iterations, a strict instability criterion of 0.0000001, 250 runs with real data and 250 runs with randomized data. Instability in all 3 runs was acceptable at < 0.0001. The preliminary runs consistently recommended three axes. Following the Peck protocol (2010), I executed the ordination manually three times using the recommended three axes, 250 runs with real data, strict instability criterion of 0.0000001, 10 iterations to evaluate stability, and 500 maximum iterations. To test whether or not chance plays a role, 250 randomizations were completed using integers selected from the computer's internal clock. Integers selected from the minutes and seconds results in unique random numbers every time the procedure is executed. These manual runs demonstrated consistency in final stress levels between 10.856 and 10.876. The ordination graphs were also qualitatively consistent. Therefore the solution was determined to be stable. The final result presented in the Results sections is the manual run with the lowest final stress, 10.856, which completed in 107 iterations. The level of final stress borders on fair per Kruskal (1964) and acceptable but cautionary by Clarke (1993).

RESULTS

Floristic Diversity

There were 111 vascular plant species (and one non-vascular species, *Marchantia polymorpha* in the Marchantiaceae family) in the peatlands (refer to Appendix 1 for complete list). Llanganuco had the highest species richness with 78 species, followed by Quilcayhuanca with 58 species, and Carhuascancha with 49 species. The 112 species are in 30 families and 12 families contain only one species. The most species-rich families are Poaceae (20 species), Asteraceae (19 species), and Cyperaceae (12 species) (Table 5.3). The most frequent species were *Plantago tubulosa* (43 plots, 66% of all plots), *Eleocharis albibracteata* (42, 65% plots), *Juncus ebracteatus* (39 plots, 60%), *Gentiana sedifolia* (37 plots, 57%), and *Calamagrostis rigecens* (36 plots, 55%). Rarity was common: 42 species occurred in only 1 plot (37.5% of all species) and 17 species occurred in only 2 plots (15.1% of all species). Peatland vegetation globally is often dominated by *Sphagnum* moss (Keddy 2010), but no species were recorded in any of the quadrats. During field work, a *Sphagnum* species was observed in small patches (< 2m diameter) near springs in the Quilcayhuanca valley. Of note is the presence of a non-native and invasive grass, *Pennisetum clandestinum*, forming dense homogeneous patches ($\geq$ ~100 m^2) in all 3 valleys. Species that indicate overgrazing were also present: *Trifolium amabile*, *Wernernia nubigena*, and *Lachemilla orbiculata*. These three species can dominate vegetation cover in overgrazed areas in the Andes (Adler and Morales 1999; Young et al. 2007).

Table 5.3. Plant families, number of species, and percent of taxa.

Family	**Number of Species**	**% of Taxa**
Apiaceae	4	3.6
Asteraceae	19	17.0
Brassicaceae	1	0.9
Campanulaceae	1	0.9
Caryophyllaceae	4	3.6
Cyperaceae	12	10.7
Equisetaceae	1	0.9
Ericaceae	1	0.9
Fabaceae	3	2.7
Gentianaceae	6	5.4
Hypericaceae	2	1.8
Iridaceae	3	2.7
Juncaceae	3	2.7
Lamiaceae	1	0.9
Lycopodiaceae	1	0.9
Marchantiaceae	1	0.9
Montiaceae	1	0.9
Onagraceae	3	2.7
Ophioglossaceae	1	0.9
Orchidaceae	3	2.7
Orobanchaceae	4	3.6
Phrymaceae	1	0.9
Plantaginaceae	2	1.8
Poaceae	20	17.9
Polygonaceae	2	1.8
Ranuculaceae	3	2.7
Rosaceae	5	4.5
Rubiaceae	3	2.7
Scrophulariaceae	1	0.9
Valerianaceae	1	0.9
Total	**112**	**100**

Reported in Table 5.4 are measures of diversity for each valley individually and all valleys combined. I calculated Whittaker's (1972) three measures of diversity, alpha, beta, and gamma. Alpha diversity is the average species richness per quadrat, or 12.6 for all valleys. The maximum species richness among all quadrats was 28 and the minimum was 5. Beta diversity, a measure of place-to-place heterogeneity in the data was 7.9 overall. Gamma diversity is the total number of species in the samples, or 112. Shannon diversity (Shannon and Weaver 1949) was 1.16, a measure that accounts for abundance and evenness. Evenness, or equitability of abundance (McCune and Mefford 2011) for all valleys was 0.46. The Simpson index, which is the likelihood that two randomly chosen individual plants will be from different species, was 0.56 (McCune and Mefford 2011).

Table 5.4. Diversity measures for all quadrats and each of the 3 valleys sampled.

		Diversity measures					
	Num. quads	**Α**	**Β**	**Γ**	**Shannon index**	**Evenness index**	**Simpson index**
Llanganuco	22	14.1	4.5	78	1.44	0.56	0.66
Quilcayhuanca	26	11.8	3.9	58	0.95	0.39	0.46
Carhuascancha	17	11.9	3.1	49	1.14	0.46	0.56
All valleys	65	12.6	7.9	112	1.16	0.46	0.56

The diversity measures reported in Table 5.4 also reveal the inter-valley similarities and dissimilarities. Llanganuco was the most species-rich and most diverse with the highest measures of alpha, beta, gamma, Shannon, Evenness, and Simpson indices. Carhuascancha, on the east side of the mountain range, was the valley with the fewest number of species. Species richness is negatively correlated with elevation, but the results were not statistically significant (Figure 5.2). Spearman's rho was -0.1371 and the p-value was 0.276. Of note is that the relationship between richness and elevation is different for each valley. Height of the tallest plant in the quadrat is a proxy for vegetation structure. Height was found to correlate positively with elevation and the relationship was statistically significant (Figure 5.3). The Spearman's rho was 0.3111 and the p-value was 0.0116. Also, here the relationship between height and elevation is different for each valley.

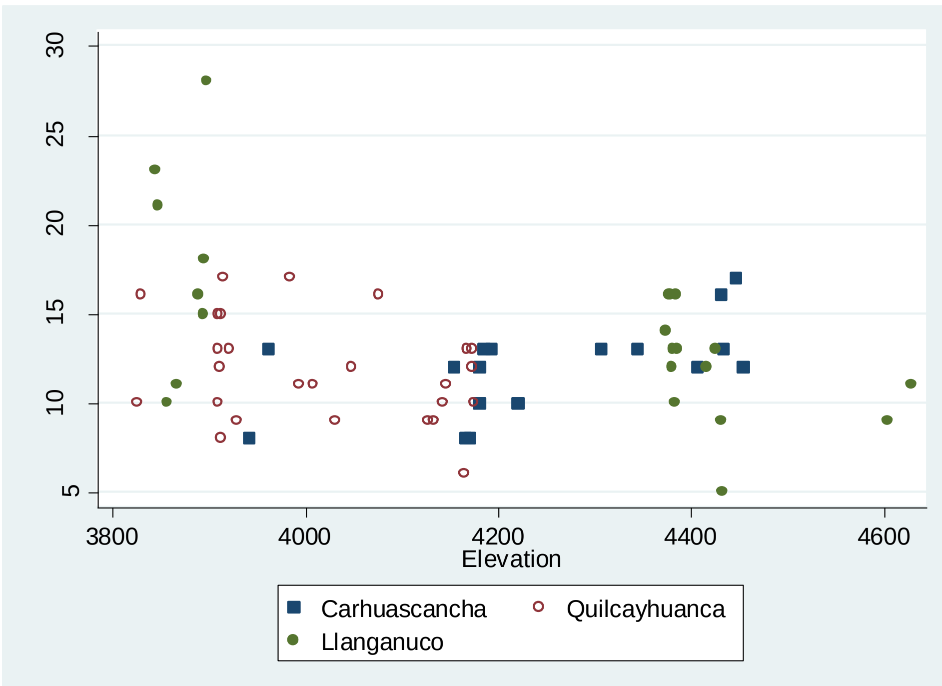


Figure 5.2. Two-way scatterplot showing the correlation between species richness (y axis) and elevation (x axis) by valley.

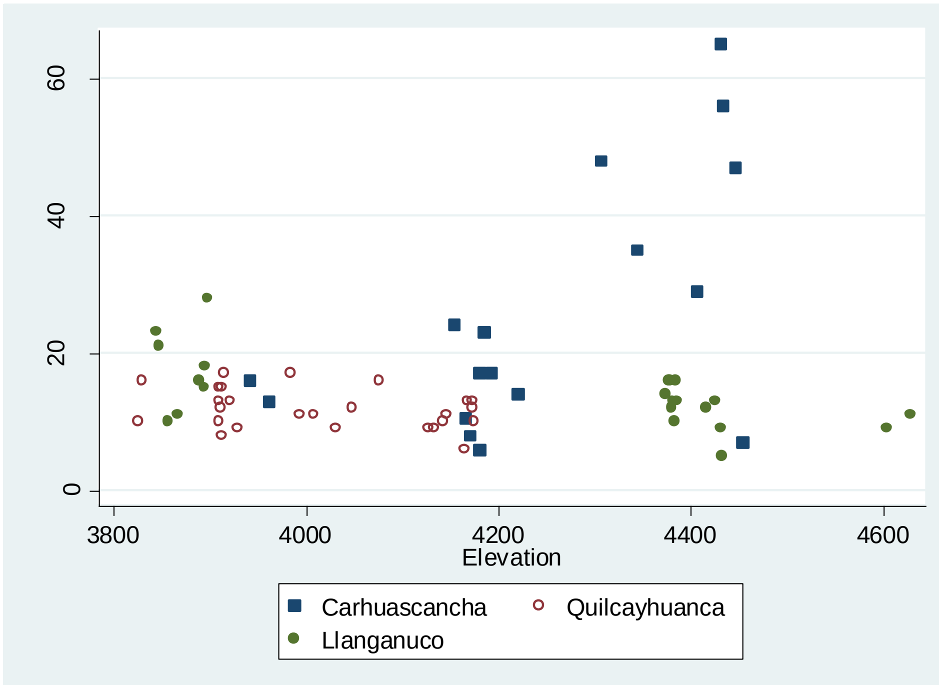


Figure 5.3. Two-way scatterplot showing the correlation between height of tallest species (y axis) and elevation (x axis) by valley.

The Steinhaus coefficient (Kent 2012), which approximates (dis)similarity among valleys, shows that the valleys are more dissimilar in their species composition than they are similar, sharing only 35% of the same species on average. Table 5.4 reports the Steinhaus coefficient for all three pairwise comparisons. The MRPP results confirm that valleys are more dissimilar than they are similar to one another then they would otherwise be expected by chance (Table 5.5). When A < 0, there is more heterogeneity within groups than expected by chance (McCune and Mefford 2011). In other words, smaller values of A indicate that there is less agreement among groups. For all pairwise comparisons, A is < 0 and p-values are statistically significant except for Llanganuco vs. Quilcayhuanca. The reasons for this lack of statistical significance are unknown, but the implication is that by chance there should be more agreement in the species richness between Llanganuco and Quilcayhuanca. For the entire dataset, the MRPP chance-corrected within-group agreement, A, equals 0.083. The p-value for the entire dataset is statistically significant at 2.1×10^{-7}.

Table 5.5. Measures of similarity among all pairwise valley comparisons.

Pair-wise valley comparison	N species shared	Steinhaus Similarity	Steinhaus Dissimilarity	MRPP A	MRPP p-value
Llanganuco vs. Quilcayhuanca	35	34%	66%	0.0077	0.1420
Llanganuco vs. Carhuascancha	32	34%	66%	0.0711	0.0000
Quilcayhuanca vs. Carhuascancha	31	37%	63%	0.0946	0.0000

Plant Communities

Cluster analysis suggests that peatland plants are organized into five groups based on dominant life form and species composition: cushion plants, sedges and rushes, sedges and grasses, and asters and sedges (Figure 5.4). The distinction among these groups was determined by trimming the cluster dendrogram at 75% information remaining representing 25% of the variation. At this level, an intuitive and interpretable affinity among species is observed. The groups are described below.

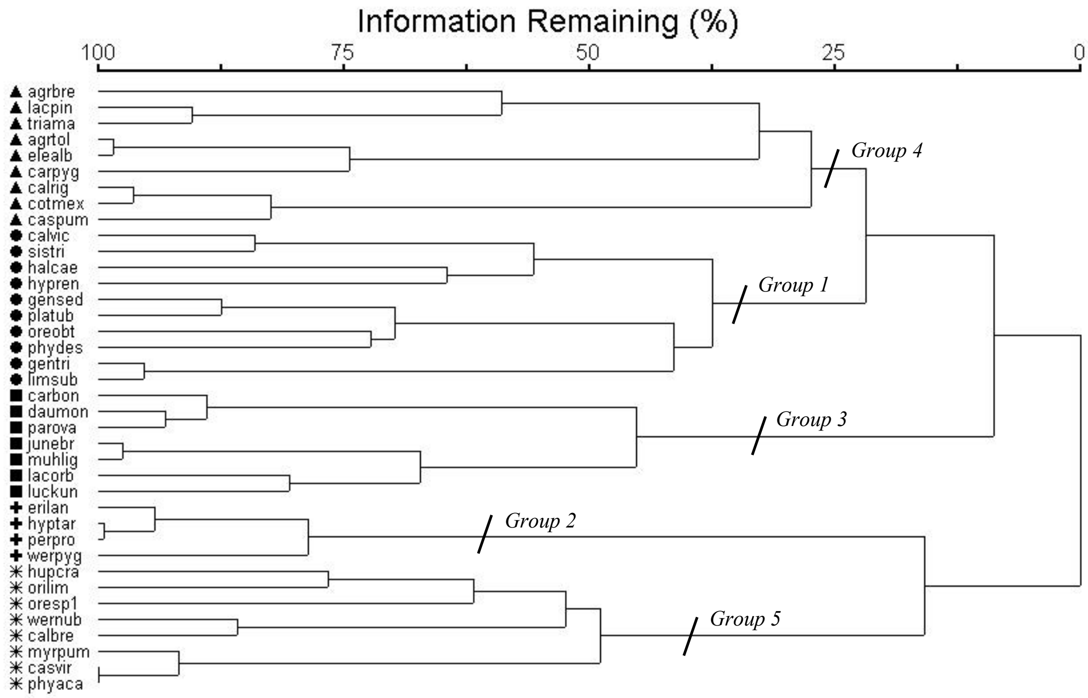


Figure 5.4. Cluster dendrogram of peatland plant communities. Symbols indicate the five species groups. The dendrogram is scaled by Wishart's objective function, which measures the information lost at each step in the analysis. Refer to Appendix 1 for species code and names.

Plantago tubulosa – Oreobolus obtusangulus, Group 1. This cushion plant group is dominated by two cushion plant species, *Plantago tubulosa* and *Oreobolus obtusangulus,* and associated species such as *Gentiana sedifolia, Phylloscirpus deserticola*, and *Hypsela reniformis* (Figure 5.5). Together, species in this community form a dense mat of vegetation that eventually develops peat. The presence of an obligate aquatic species, *Limosella subulata*, indicates that soils are saturated year-round (León and Young 1996).

Figure 5.5. The *Plantago tubulosa – Oreobolus obtusangulus* Group 1.

Werneria pygmaea – Pernettya prostrata, Group 2. The second cushion plant group is dominated by *Werneria pygmaea*, a peat-forming cushion plant. Interspersed in this cushion community is *Pernettya prostrata*, a species with red berries that is important to wildlife (Troya, Cuesta, and Peralvo 2004). Less abundant but typical of the community are two Asteraceae species, *Erigeron lanceolatus* and *Hypochaeris taraxacoides*, which occur in bogs and seeps. Photograph is unavailable.

Juncus ebracteatus – Carex bonplandii, Group 3. This sedge-rush community is one of the most characteristic groups in the study area dominated by the abundant short sedges (5-10 cm in height), *Juncus ebracteatus* and *Carex bonplandii* (Figure 5.6). The rosette-shaped *Lucilia kunthiana* is also common in this community.

Figure 5.6. The *Juncus ebracteatus – Carex bonplandii* Group 3.

<u>*Eleocharis albibracteata – Calamagrostis rigecens – Lachemilla pinnata*, Group *4*</u>. This sedge-grass community is dominated by *Eleocharis albibracteata*, a sedge that is often accompanied by the herbaceous *Lachemilla pinnata* in montane grasslands with moist soils in valley bottoms (Cabido, Breimer, and Vega 1987; Funes et al. 2001). The grass, *Calamagrostis rigecens*, is also common in Peruvian mountain peatlands (Flores, Alegría, and Granda 2005). Of note is the abundant legume in this group, *Trifolium amabile*, an indicator of livestock impacts. Photograph is unavailable.

<u>*Werneria nubigena - Oritrophium limnophilum – Huperzia crassa*, Group *5*</u>. This aster-sedge-club moss group is dominated by large mats of *Werneria nubigena*, a basal rosette that spreads when overgrazing occurs (Young et al. 2007) (Figure 5.7). The sedge *Oritrophium limnophilum* characterizes this community along with *Huperzia crassa*, a species from the Lycopodiaceae family that requires moist soil to survive.

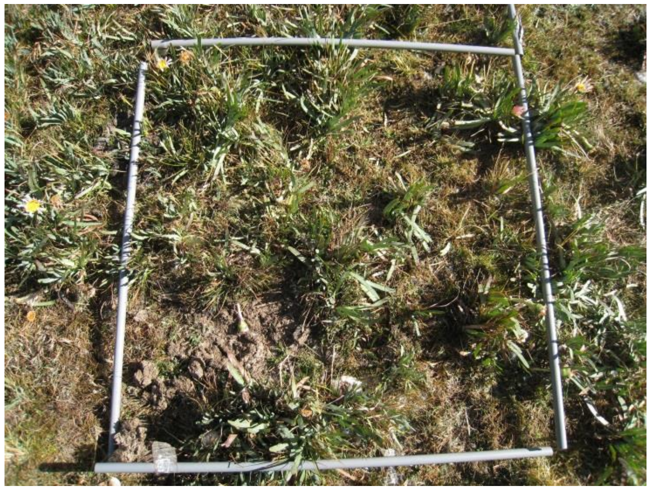


Figure 5.7. The *Werneria nubigena – Oritrophium limnophilum – Huperzia crassa* Group 5.

Controlling Factors

The abiotic factors statistically controlling peatland vegetation composition were elevation and a combination of bulk density, percent organic matter, and CEC (Figures 5.8, 5.9, 5.10). The three axes produced by the NMS analysis together represented 87% of the variance. Axis 1 represented 35% of that variance and is associated with elevation. Axis 2 represented 19% of the variation, although its environmental association is unclear. Axis 3 represented 33% of the variation and appeared to be associated with a combination of bulk density, OM, and CEC. Because Axis 2 is unexplained, I re-ran the NMS using 2 axes to determine whether or not a solution with lower stress would result. The 2-axis solution stress in relation to dimensionality was 16.845 versus 10.586 from the 3-axis solution; therefore the 3-axis solution is more reliable. The quadrats did not demonstrate a strong pattern of differentiation when grouped by valley, wetness index, or pH. Presented in Figures 5.8, 5.9, and 5.10 are the joint bi-plots showing the association between the quadrats and elevation, bulk density, percent organic matter, and CEC grouped by valley, wetness index, and pH quartiles. Abiotic vectors with an $R^2 \geq 0.2$ are displayed.

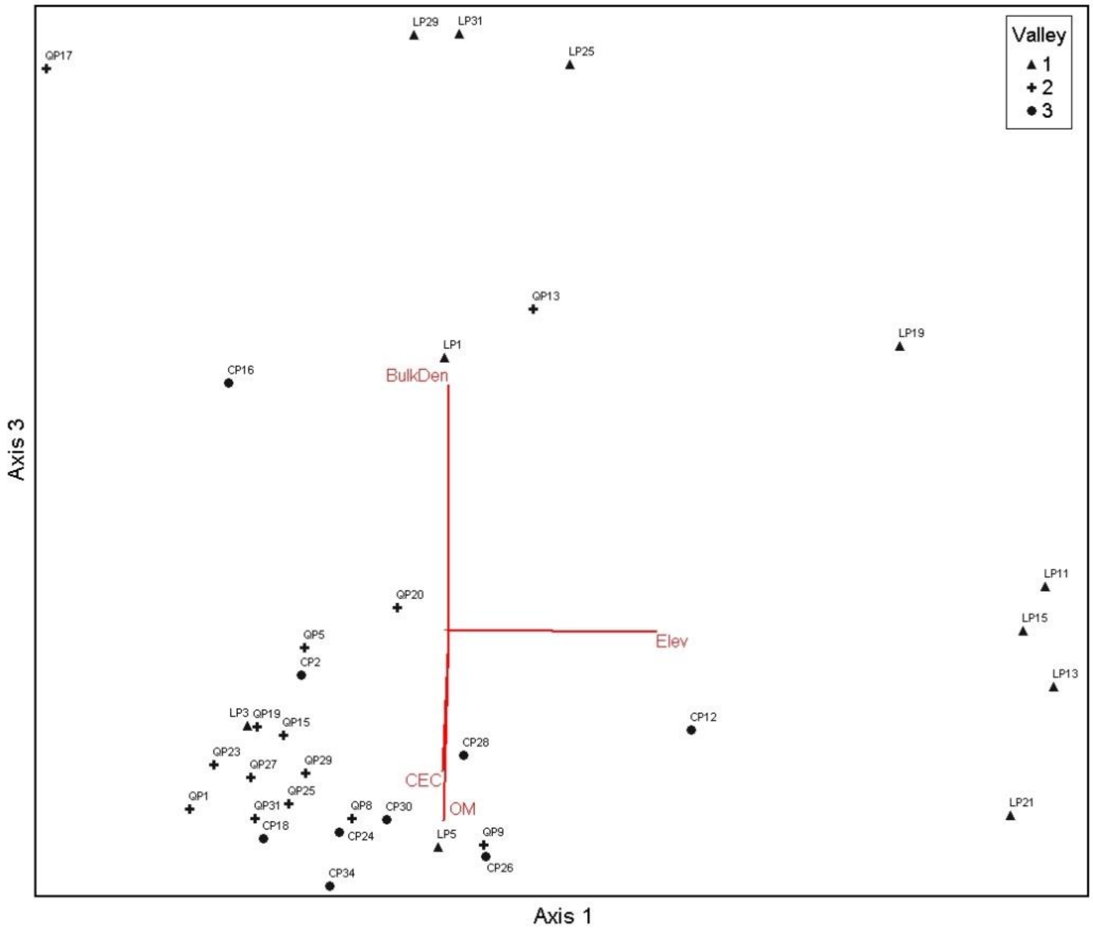


Figure 5.8. NMS ordination of quadrats in peatland species space with joint bi-plots of % organic matter (OM), cation exchange capacity (CEC), bulk density (BulkDen), and elevation (Elev) relative to Axes 1 and 3. Symbols correspond to valleys: ▲= Llanganuco (1), + = Carhuascancha (2), ● = Quilcayhuanca (3). Displayed vectors have $R^2 \geq 0.2$.

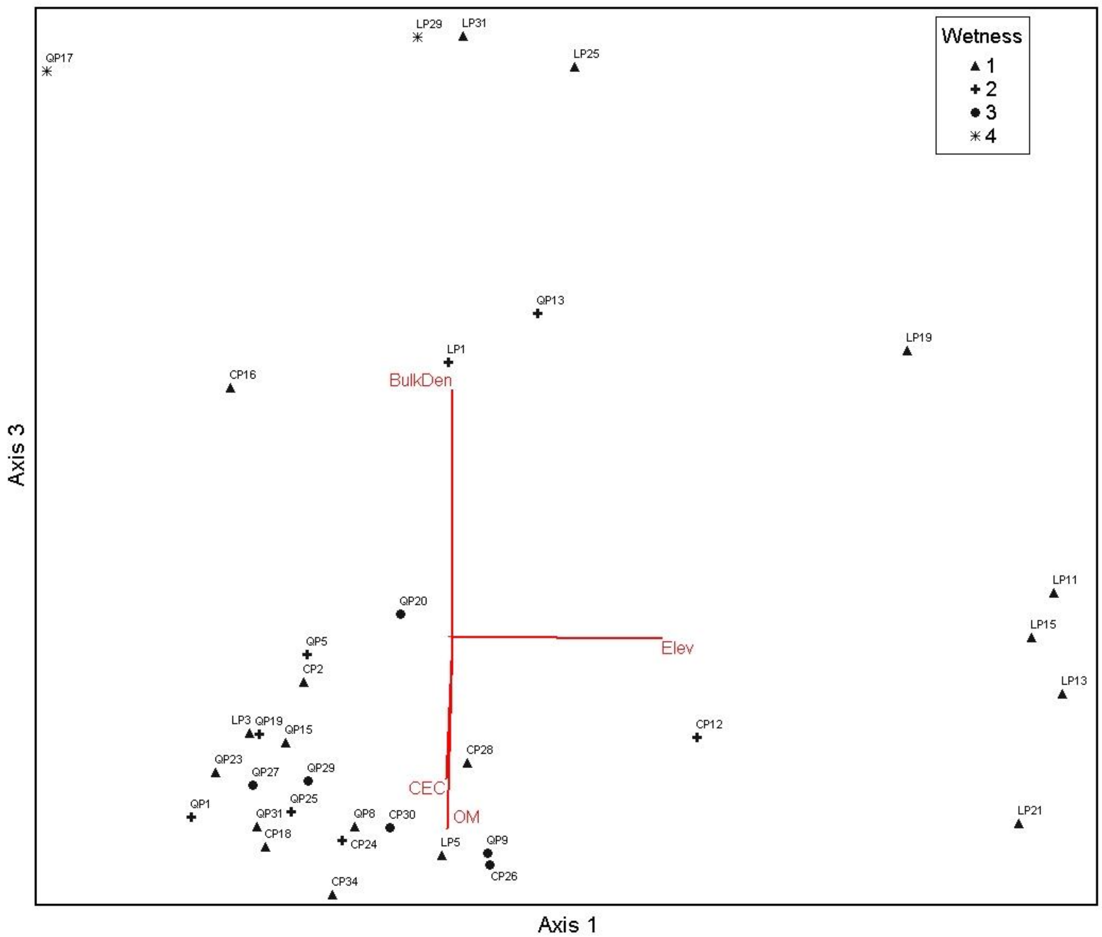


Figure 5.9. NMS ordination of quadrats in peatland species space with joint bi-plots of % organic matter (OM), cation exchange capacity (CEC), bulk density (BulkDen), and elevation (Elev) relative to Axes 1 and 3. Symbols correspond to wetness index: ▲ = hummocks saturated soil, no standing water (1), + = saturated soil, no standing water (2), ● = <25 cm standing water (3), ✳ = seasonal peatland (4). Displayed vectors have R^2 ≥0.2.

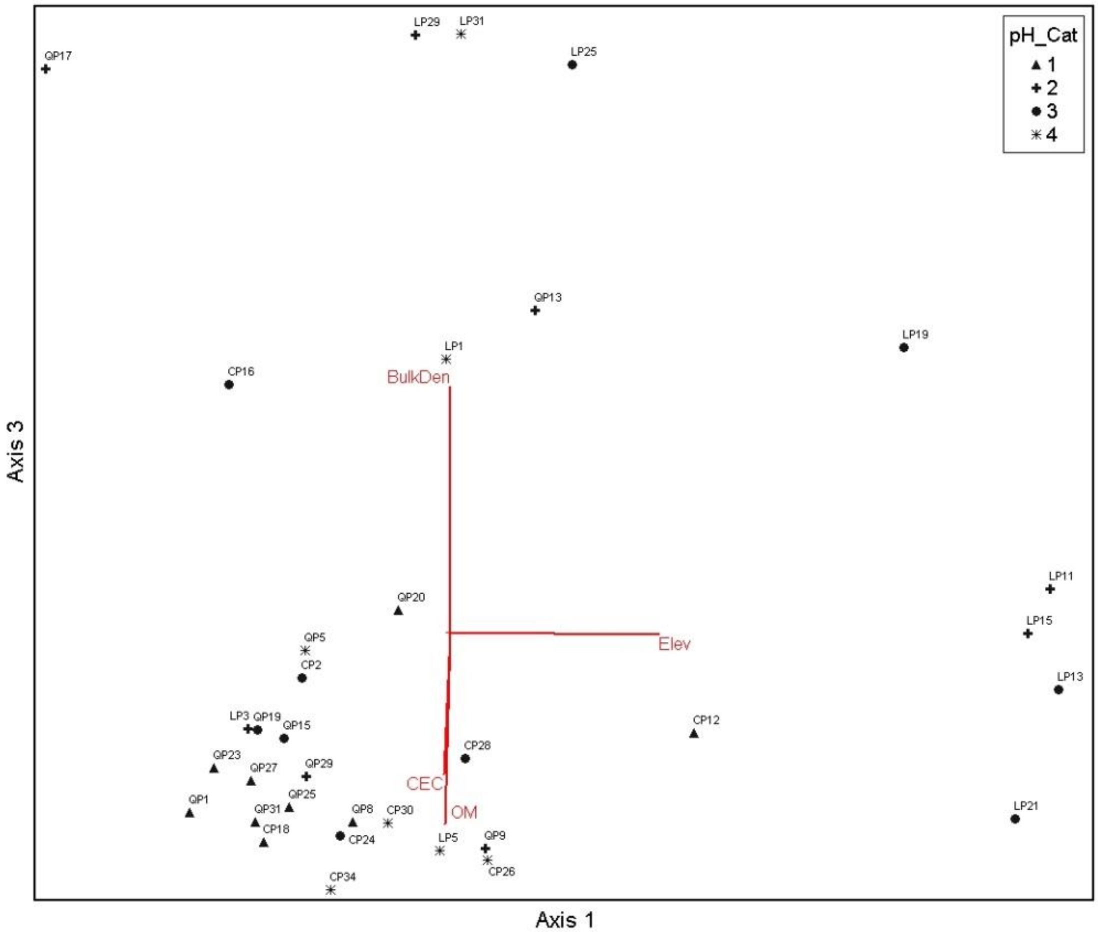


Figure 5.10. NMS ordination of quadrats in peatland species space with joint bi-plots of % organic matter (OM), cation exchange capacity (CEC), bulk density (BulkDen), and elevation (Elev) relative to Axes 1 and 3. Symbols correspond to pH quartiles: ▲= 3.66 - 4.07 (1), + = 4.08 – 4.50 (2), ● = 4.51 - 4.67 (3), ✳= 4.68 – 5.36 (4). Displayed vectors have $R^2 \geq 0.2$.

DISCUSSION AND CONCLUSION

Relative to flora in the mountains of East Africa and New Guinea, the Andes have the highest total plant diversity (Sklenář, Hedberg, and Cleef 2014). This study provides additional evidence to support the high plant species diversity in the Andes because HNP peatlands have high species richness and diversity. The intra-valley comparison implies that there are probably more plant species present in the peatlands than were identified during fieldwork and follow-up by botanists. Because only ~33% of species were shared between pairs of valleys, sampling additional valleys in the mountain range would most likely add more species to the known peatland flora. Bigger valleys with bigger peatlands would be expected to contain more species because the species richness-area relationship (correlation between species richness and the log-transformed geographical area of the mountains) is strongly correlated in comparison to the same relationships in the mountains of East Africa and New Guinea (Sklenář, Hedberg, and Cleef 2014). In the Peruvian *jalca* ecosystem, Tovar and others (2012) used the same 4 m^2 quadrat sampling scheme and reported mean alpha diversity of 18 compared to 12.6 in my study area. They also reported a mean Shannon index of 2.3 versus 1.16 in this study. While the diversity measures reported herein are lower than those of Tovar and others (2012), further sampling and the addition of species to the taxa would increase the diversity levels. The species list reported here enhances the work of the Missouri Botanical Garden and Smith's (1988) dissertation, but more botanical sampling should be undertaken to create a comprehensive peatland taxa. For example, bryophytes and lichens are important in these ecosystems (Cooper et al. 2010), but were not sampled in this research and represent a future research prospect. To understand the effects of climate change on peatland vegetation, monitoring taxa and measures of diversity and richness over the long-term should be a priority. Future measurements could be compared with those reported in this study, which effectively serve as a baseline.

The plant communities found here resemble the qualitative observations made by Smith (1988, 43) and specify quantitatively that cushion plants are associated with species in the Cyperaceae, Asteracea, and Poaceae families. Whereas Weberbauer (1945) and Maldonado-Fonkén (2015) present four hydrophitic peatland communities (*Distichia* peatland, peatland with mosses and shrubs, peaty meadow, and stream grassland), this research shows that peatland vegetation communities display more heterogeneity than previously thought and may merit finer levels of categorization. For example, the communities identified in this research could be conceived as sub-types of peaty meadows.

Of importance is that the cluster analysis suggests that some plant communities are impacted by grazing cattle, sheep, and horses. For example, *Werneria nubigena*, an indicator of overgrazing and unpalatable to non-native livestock (Young et al. 2007), is the dominant species in the *Werneria nubigena - Oritrophium limnophilum – Huperzia crassa* community. Based on field observations and supported by abundance data (present in 23 of 65 plots), *Werneria nubigena* dominates some peatland patches. Informal conversations with local pastoralists confirmed that the species has been expanding spatially. Vegetation cover and peat soil are disturbed by herbivory, hoof action and other destructive activities (cf. Pérez 1993), creating conditions in which *Werneria nubigena* can colonize, expand, and begin to dominate plant community composition. *Werneria nubigena* contains pyrrolizidine alkaloids, a compound poisonous to humans and livestock (Roeder, Bourauel, and Theisen 1992; Roeder and Pflueger 1995; Bildfell 2013). As another example, *Trifolium amabile* was a dominant species in the *Eleocharis albibracteata – Calamagrostis rigecens – Lachemilla pinnata* community and is an indication of overgrazing, although it is highly palatable (Adler and Morales 1999). Seeds from *Trifolium amabile*, a legume, are in part dispersed by livestock.

The cluster analysis provides evidence that livestock grazing could similarly be affecting peatland vegetation composition. Although grazing impacts were not measured, *Trifolium amabile* and *Werneria nubigena* are abundant species in 2 plant communities that are related to overgrazing as mentioned above. Experimental approaches in the Venezuelan and Ethiopian highlands have shown that cattle grazing affects not only vegetation organization, but also hydrology (Molinillo and Monasterio 1997; Mwendera and Saleem 1997). In this analysis, there was no statistically significant factor attributable to Axis 2 (19% of the variance). One possible explanation for the unexplained variance in Axis 2 is grazing, at least partially. Axis 2 could also be a combination of factors that control wetland vegetation composition that were not measured. For example, groundwater chemistry was identified by Cooper and others (2010) as a controlling factor. In their case, the combination of pH and HCO_3 were significant. Determining whether or not grazing is a controlling factor (or part of a combination of factors) would require further research.

Mountains are characterized by steep environmental gradients, meaning that abiotic factors change quickly over short distances. Elevation is a well-recognized variable that is comprised of interrelated environmental gradients that organizes mountain vegetation composition (Körner 2003, 2007). Along the elevation gradient, precipitation and temperature patterns vary and thus drive vegetation, although local conditions such as nutrient availability and disturbance history are also important factors (Hadley, Price, and Grabherr 2013). In the Peruvian *jalca* grasslands, Tovar and others (2012) found that elevation and soil pH were the most important gradients that controlled species composition. Therefore the Axis 1 association with elevation is unsurprising; however in this study, pH was surprisingly not a factor although it is important elsewhere (Cooper et al. 2010; Tovar et al. 2012). Overall pH values ranged from 3.6 – 5.4 with a mean of 4.4, or acidic. These values are substantially lower than the values reported by Cooper and others (2010) (3.2 – 8.2) and Salvador and others (2014) (4.5 – 7.9). Peat soils typically have low pH values and high CEC values (Zinck 2011; Rydin and Jeglum 2013) and in this case, CEC was shown to more strongly influence peatland vegetation.

CEC and OM together with BD accounted for 33% of the variation on Axis 3. As peatland soils decompose, organic acids, lignins and other compounds are produced that then increase CEC and decrease pH levels (Rydin and Jeglum 2013). So as OM content increases, CEC also increases and pH is low. Vectors on the joint biplot show the opposite or inverse relationship between OM and BD. When soil OM is high there is more pore space among soil particles, thus BD is low. In this study, mean OM was 46% and median was 55%, whereas mean BD was 0.3 g/cm^3 (300 kg/m^3) and median was 0.1 g/cm^3 (100 kg/m^3) (Table 5.2). Organic matter and BD can be used as indicators of decomposition and the soil samples taken in this study point to moderate to strongly decomposed. Peat soils with BD <75 kg/m^3 are fibric, or weakly decomposed. BD > 75 kg/m^3 and < 195 kg/m^3 are mesic, or moderately decomposed. BD > 195 kg/m^3 is humic, or strongly decomposed (Letts et al. 2000; Rydin and Jeglum 2013). The minimum BD is 100 kg/m^3, maximum is 1200 kg/m^3, and the median is 100 kg/m3 indicating a moderate to strongly decomposed peat. Soil samples were collected at 10 cm depth where plant roots are influenced by chemical and physical soil characteristics such as BD and associated levels of decomposition along with CEC and OM.

Relative to peatlands in other areas of Peru that have been examined by researchers, this study shows that HNP peatlands are unique and merit further scientific and special attention from land managers. Peatland in HNP are highly acidic, especially in comparison to pH levels in similar systems (Cooper et al. 2010; Salvador, Monerris, and Rochefort 2014), but this analysis did not show that pH is a controlling factor of vegetation composition. Instead, CEC, OM, BD, and elevation are the important factors controlling vegetation composition. Plant communities demonstrate high heterogeneity and valley-to-valley dissimilarity, implying that biological diversity inside HNP may be higher than previously estimated. Peatlands in the Peruvian Andes face varied threats (Salvador, Monerris, and Rochefort 2014) and among these, this study points to the effect of non-native livestock as an agent of change in the vegetation ecology of HNP peatlands. To aid land managers in administering policies that protect the future of peatlands, further work is necessary to quantify the impacts of cattle, sheep, and horses on these unique and biologically diverse high Andean ecosystems.

Chapter 6: "They Are Drying Out": Ecological Oral Histories of Mountain Peatlands

INTRODUCTION

The tropical Andes have been inhabited for millennia and the Cordillera Blanca and surrounding areas are no exception (Ellenberg 1979; Lynch et al. 1985; Gade 1996; Contreras 2010). Such long-term land use has produced landscapes that exhibit legacies of both human and biophysical processes and the effects of feedbacks between those processes (Young 2008, 2009). Because mountain landscapes are the result of coupled natural and human influences, SES theory lends itself well as a framework for evaluating the consequences of climate change. Prior chapters addressed aspects of the biophysical systems including spatio-temporal change, driving factors of change, biological diversity, and explanatory abiotic variables of vegetation heterogeneity. In this final concluding chapter, I turn to the equally important social dimensions of peatlands in HNP and ask the following three questions: 1) How do local people perceive peatland change? 2) Do local perceptions corroborate quantitative findings presented in previous chapters? and 3) Does a mixed methods approach enrich the narrative of peatland change?

These questions are addressed using a method called ecological oral history, a qualitative technique that documents landscape and ecological changes of a particular place over time. In contrast to the methods used in previous chapters that rely on quantitative epistemologies of geospatial and multivariate analysis, ecological oral histories document change in a way that is sensitized to the nuances of people's relationships with natural resources, and in this case, peatland landscapes. Landscape change is evaluated "from the perspective of people who have used that land over time and are intimately embedded within people's life experiences" (Nightingale 2003, 81).

Applying SES theory to understand and evaluate place-based change, as is the case in this dissertation, requires integrated approaches and methodological pluralism. Assessing and constructing landscape histories and legacies in rural areas affected by climate-induced glacier recession like HNP necessitates the integration of both social and ecological methods because these landscapes are "social constructs and biophysical entities" (Duncan, Kyle, and Race 2010, 432). For large projects covering sizeable geographic areas or ecosystems such as the ones studied by Collins and colleagues in the Long Term Ecological Research network (2011), a team structure comprised of specialists from multiple disciplines is ideal. The PPD model is a systems-based expression of human-environment interactions and coupled natural human systems paradigms that have long been at the heart of geographic thought. Furthermore, with its its integrative strategy, the PPD model is reminiscent of a recent call by Lave and others (2014) for geographers to engage in "critical physical geography" in order to better understand landscapes that are the co-production of biophysical and social processes. Advocates propose that critical physical geography (CPG), should be a new subfield of geography combining "critical attention to relations of social power with deep knowledge of biophysical science or technology in the service of social and environmental transformation" (Lave et al. 2014, 7). In other words, human geographers should not only engage more with biophysical processes relevant to their study areas, but also understand how these processes influence human agency and inequality. Simultaneously, physical geographers should grasp the political and social issues that influence their positionality and study areas. For geographers working in the realm of landscape change, to ignore the diversity and contributions of either social or biophysical processes would be short-sighted (Lave et al. 2014). I contend that applying the PPD model in HNP is to implement CPG. Because the landscapes inside the park boundaries are the product of both biophysical and social processes, addressing the social dynamics of peatlands is necessary.

Prior chapters have exclusively dealt with geospatial methods, relying on remote sensing, GIS, landscape metrics, spatial statistics, and multivariate ecological analysis. These methods produce knowledge about landscapes based on quantitative analytics, or the positivist epistemology of quantitative methods (Mattingly and Falconer-Al-Hindi 1995; Nightingale 2003). For example, classification of six Landsat TM images and the subsequent pattern metrics analysis allowed me to link pattern and process over time and thus infer aspects of ecosystem functionality and integrity. The method produces knowledge about the extent and character of wetland change over time, or a chronotopic dataset about 23-year history of peatlands inside the park. Conclusions about alpha and beta diversity, the effect of elevation on vegetation structure, and the influence of abiotic factors on vegetation composition are based on extensive field campaigns that create vegetation datasets that were then evaluated quantitatively to draw conclusions about heterogeneity in the park.

These analytical techniques, while proven, necessitate expert training and advanced analytical skills. Writing about the methods she used to evaluate nature-society issues in community forestry projects in Nepal, Nightingale (2003) recognized that using aerial photos to map forest cover change is an epistemology that requires specialized knowledge and expertise to interpret. Drawing on Haraway's (1991) concept of the Cartesian view from no-where that criticizes remotely collected imagery as situated, partial and privileged, Nightingale (2003) opts to incorporate ecological oral histories to overcome this privileged perspective and add an enriched narrative about community forests. Nightingale (2003) uses mixed methods and a feminist framework to intentionally interrogate the biases inherent in knowledge produced from differing epistemological traditions; she focuses on the inconsistencies between the aerial photographs and the ecological oral histories. In doing so, Nightingale (2003) finds that the process reveals more insight about social and ecological processes that transform forests than would have been possible without methodological pluralism. In a similar vein, Jiang (2003) adds remote sensing to an ethnographic approach to help evaluate human perceptions of landscape change. She seeks to synthesize otherwise disparate epistemologies to develop deeper insight into land use, landscape perceptions, and implications of cultural change in pastureland in Inner Mongolia, China. Also working in and around HNP, Lipton (2008) used used mixed methods – remote sensing and interviews – to characterize landscape change. Her analytic foundation was political and cultural ecology that offered insights into the spatial and temporal context of climate change, access and conflict over resources, and conservation. The research presented in this chapter is influenced by methodological pluralism utilized by Jiang (2003), Nightingale (2003), and Lipton (2008) and focuses on the social dimensions of peatland change, but situates the work within SES theory. One primary difference is that this research tests the application and utility of a specific device, the PPD model (Collins et al. 2011) and makes recommendations for refinement. Furthermore, what differentiates this chapter from other works (Jiang 2003; Nightingale 2003; Lipton 2008) is that it responds to the recent call for geographers to engage in CPG (Lave et al. 2014). Whereas Lipton's interviews inquired about land tenure, landscape-scale changes, socio-political history, and national park sensitivities, the interview results in this chapter report on perceptions about peatland changes. Doing so contributes to our understanding of local knowledge about the secondary effects of climate-related glacier recession in tropical mountains because it delves into the social dimensions of an ecosystem that is not well understood.

METHODS

Ecological oral history augments quantitative records of landscape change by including "traditionally excluded and ordinary people who experienced the historical phenomenon in question" (Endres 2011, 485). The method is also known as environmental oral history (Endres 2011), but here I use the term ecological oral history following in the tradition of geographers who use it to document changing landscapes (Nightingale 2003; Hanson 2015). An advantage of analyzing ecological oral histories together with other quantitative methods is that it provides additional information that is otherwise not provided, thereby gaining insights (Duncan, Kyle, and Race 2010). These insights come from guided conversations with subjects who have extensive knowledge of the landscape that they have amassed through ongoing experiences with specific places.

According to Fogerty (2005), there are criteria required for oral histories to be credible. The study should be designed thoughtfully to collect relevant and meaningful information through multiple interviews. The histories should be useful beyond the immediate needs of the interviewer. In other words, the histories serve a greater purpose than simply recording recollections. Ecological oral histories record local knowledge, defined as a "dynamic system of place-based observations, interpretations, and local preferences that inform people's use of and relationship with their environment and with other people. It may include a mix of social, ecological, and practical knowledge and involve a belief component" (Chapin, Kofinas, and Folke 2009, 348). For this study, local knowledge is used to corroborate findings derived from disparate methods and enrich the narrative of peatland change in HNP.

Forty-two semi-structured interviews were conducted lasting no more than 1 hour. Approval from the University of Texas Institutional Review Board was granted on June 6, 2012 for a 3-year period (protocol number 2012-01-0009; see Appendix 6 for IRB Approval Letter). Individual participants' privacy is protected and names included herein are abbreviated to the first initial of the participant's last name. The sampling strategy targeted people who are or have been in direct contact with peatlands, either through their professional occupation (i.e. pastoralists) or indirectly through policy applications (park staff). Participants represent various stakeholders and beneficiaries of peatland ecosystem services: *arrieros* (donkey drivers), pastoralists, tourists (including alpine climbers and hikers), mountain guides, tourist agencies, NGOs, regional and national officials, engineers, and consultants. The subjects have extensive knowledge of the landscape that they have amassed through ongoing experiences inside the park. Most pastoralist families have been grazing in the park for 4 or more generations and ecological knowledge is passed down through generations. Given this family history and that older people (> 55 years of age) have amassed longer experiences inside the park, there was some selection bias towards older participants. Pastoralists and tourists were selected randomly by approaching subjects walking inside park boundaries. Pastoralists were interviewed either a) on Saturdays and Sundays when the owners are herding, checking on animal health, and verifying that all animals are accounted for; or b) following community meetings. As a consequence, the sampling strategy omitted any potential participants that were not present in the valleys on Saturdays and Sundays and those who were not present at community meetings. All others were selected through snowball sampling where subjects were asked for two names of other potential subjects. Interviews were conducted between July and December 2014. Of the 42 participants, 8 were females and 34 were males and all were over 18 years of age (stipulated by the IRB). Ages ranged from 18 to 75. Nine were documented through written notes and 33 were recorded with a hand-held digital voice recorder. Interviews were completed with the assistance of Gladys Jiménez, a native of the area who is fluent in Spanish, Quechua, and English. Ten were completed by the author, 29 by Ms. Jiménez, and 3 jointly by the author and Ms. Jiménez. Quechua to Spanish translations were completed by Ms. Jiménez and Spanish to English translations were done by the author.

Interviews with pastoralists and some tourists took place in the valleys and before or after community meetings. Interviews with officials, NGOs, tourists and others took place in homes, offices, and coffee shops in Huaraz. These locations were chosen intentionally to create an encouraging climate for conversation in the subject's "home turf" (Fogerty 2005, 109). After receiving verbal consent, 16 questions were used to structure the interview (Table 6.1). They are intentionally open-ended so as to capture knowledge rather than guide answers. The questions were presented in Spanish, Quechua, or English depending on the participant's preference. After all the interviews were collected, analysis was performed by reviewing the recordings and organizing key quotes and insights thematically.

The tone of the interviews was intended to be pleasant, engaging, respectful and non-judgmental. In terms of personal positionality, I made every effort to remain unbiased in my observations. There were issues of distance between me and the participants that imply limits on my ability to fully capture all of the participants' perceptions. One factor that created distance is language. I am fairly fluent in Spanish, but I do not always comprehend all the nuances in every conversation. Additionally, Quechua words often appear in conversations and I have no knowledge of Quechua. In cases where Ms. Jimenez conducted the interviews in Quechua and translated them to English, the distance between me and the participants was even greater. The interviews were designed to capture perceptions of spatial and ecological peatland change, but they do not encapsulate the full range of participant experiences and household and community dynamics that can be collected when the investigator spends one year or more in the study area. These "deep hanging out" experiences (Gelles 2000, xii) permit the investigator to collect hundreds of interviews and become intimately involved in community activities of all kinds. In contrast to the year-long or more experiences that are comprehensive, the ecological oral histories focus on a singular ecosystem in the context of glacier recession.

Table 6.1. Guiding questions for the ecological oral histories.

- How are peatlands used?
- Have you noticed any changes in the peatlands? Compared to when you were a child?
- After showing participant a photograph of *Plantago tubulosa* (*zampa*), interviewer asks: do you recognize this plant? Is there more or less of this plant compared to the past or to when you were a child?
- Do you think the peatlands have been impacted by glacier recession?
- Have you noticed any changes in water quality or quantity in the valleys (*quebradas*)?
- What is the source of water in the peatlands?
- Do you receive a benefit from the peatlands?
- Does grazing affect the peatlands?
- Do you know any myths or legends about the peatlands?
- Do you make any offerings to the peatlands?
- Does your community have a program or policy for using the peatlands?
- What is your opinion of the peatlands?
- Do the peatlands have a unique or important characteristic?
- How will the peatlands look in the future, 5 or 10 years from now?
- How do you want the peatlands to look in the future, 5, or 10 years from now?
- Are peatlands a good place to camp?

RESULTS

On the whole, perceptions of local people corroborate quantitative findings reported in previous chapters. Observations documented in the interviews indicate that peatlands are decreasing in area and becoming drier, changes that are accompanied by ecological impacts, decreasing water quality and quantity, and a noted loss of beauty.

Among participants there was little consensus among causes driving peatlands change and sources of peatland water. The following themes were generated from major concepts revealed in the interviews.

Several questions did not yield useful answers and are therefore not included below. For example, there were no myths or legends about peatlands recounted, nor were there any stories about offerings. The answer to these questions was always definitively negative. The two questions about the future of wetlands were ineffective at producing any visions or ideals over the next 5 to 10 years. Overall, interviews with tourists were not useful because of their limited experiences in the study area. Most had only been to the area once or twice at most in the past so they have no historical reference point or sense of change. Tourists who are mountaineers or alpine climbers do not seem to pay attention to vegetation on their way up to high altitude base camps.

Changes in Size, Wetness, and Aesthetics

Among participants, the general consensus was that peatlands are losing area. Only 2 participants stated that they have not seen any changes, whereas all pastoralists, *arrieros*, park officials, and NGO staff indicated that peatlands are becoming smaller. Phrases that were commonly used to describe changes in extent included getting smaller, drying out, shrinking, and disappearing. J., an *arriero*, noted that "peatlands are smaller and there's less water in the rivers." H. offered more detail: "we are observing that the *oconales* were larger. There were a lot of them. Now we see that there are few. It's not like before. I think, every year, they are drying out 20 to 30 meters. That's how they're getting smaller. Now there aren't enough." Reflections offered by park officials, NGO employees, and guides agree that peatlands are getting smaller, but their comments were less descriptive overall. Conversations tended to lean towards solutions about what could be done to prevent loss of area and restoration efforts. Observed changes in wetness were also common among participants. M., an engineer who started working in the park in the 1960s, said that "before, during the dry season, you couldn't get across [the bofedales] even on a horse…we used to stick sticks in the bofedales and water came out, but now it doesn't come out." R., a *llavera* (a *llavera* (masculine *llavero*), translated as keyholder, is a position of trust and responsibility elected by the community. The *llavera* controls access into and out of the park at a community-controlled gate inside the legal park boundary by unlocking or locking a padlock) and community leader noticed inter-seasonal differences. During an interview that took place in the height of the dry season she said "usually [bofedales] are a little wetter right now." Changes in spatial configuration and fragmentation were also noted. For example, H. said "there are pieces [of *oconales*] now. There were more, but now they are drying out." Three participants made comments about the changing aesthetics of the peatlands saying they "used to be prettier."

Ecological Changes

Accompanying changes in peatland size, wetness, and beauty, participants also noticed ecological differences. These observations were focused on organisms, both plant and animal. Four participants reported that frogs have vanished. R. said that "before there were frogs. You could hear them all over the place, at night and during the day…kwok, kwok, kwok…and now, no." D. remembered "grasses now aren't like they used to be, there is less. There are animals in danger of extinction. The last time I saw a frog was 10 years ago. They were big and black frogs. They were big and pretty. And now you don't see them…you don't see them. What has happened? That's the question." The species is not known at this time, but one possibility could be *Telmatobius mayoloi*, an endangered frog endemic to Ancash that is harvested for food and medicinal properties (Icochea and Lehr 2004). Effects on trout and livestock were also reported. Trout die and cows and horses seem to be skinnier and sicker. E. said "the grasses are different. Now the grass is not like it was. It has been reduced because horses eat a lot, more than cows. Higher up, in Cuchillacocha, there were trout, but now, no." When shown photographs of *Plantago tubulosa* (locally called e*strella, estrellita,* or *zampa*), a small rosette and one of the most common species in the peatlands, participants say it is now scarce. R. recalled that "years ago, there was more, it was replete, fuller. This plant was all over, but now not really." C., referring to *Plantago tubulosa* using Quechua said "Even now I see plants that are native to the wetlands like *qachqa uqu*, but they are drying up." Local communities are also changing the ecology of bofedales by draining them via canals or ditches that redirect water where it can be used in agricultural fields. D., in Quechua, explains that the communities dig canals "because the water doesn't get to the places we want it to. We take the water from the wetlands to irrigate dry areas."

Water: Sources and Changes in Quantity and Quality

When asked about the source of water in the peatlands, responses varied widely, but mirrored the sources of hydrologic input presumed by researchers: from glaciers, filtration, subterranean channels, rainfall, snowfall, and mountain peaks. In contrast, comments about water quantity were unanimous in that there is now more scarcity. Several participants reported less water overall in the valleys, streams, and rivers. D., a pastoralist, remarked that there "used to be a lot of waterfalls all year long, but now it's just drops." Water scarcity is discussed along with less glacier ice and comments about water quality. Both L. and D. said that there is a direct connection between peatlands, plants, and water quantity and quality. According to D., "this water will make you sick to your stomach. This water here, no [pointing to river], but from above it's fine to drink…trout that come into this river die because it's not good. It makes cows skinny and sick. They die and maybe the water makes them sick." Similarly, J. observed that "there are more fungi and viruses in the water. If you drink it, you'll get sick."

Causes of Peatland Change

Participants' reasons for what could be causing peatland change were inconsistent. Numerous causes varied from participant to participant and included animal grazing, mines, the 1970 earthquake, climate change, the global phenomenon, lack of caring for peatlands, littering, lack of water, and glacier recession. Participants commonly interconnected glacier recession, water quantity, and ecological changes. H. identified "the global phenomenon", which is most likely a reference to climate change and which he linked to observed ecological changes: "the global phenomenon affects *oconales*. The sunlight has changed completely and it affects the grasses. Also, lack of caring for them, lack of water. The water is not getting there anymore. That's why the *oconales* are drying out." M. said that "animals – pigs, horses, cattle, sheep, donkeys – have been messing up [bofedales] for a long time…less glacier ice is causing drier wetlands. Now it's just rainfall." Mining was blamed as a factor in peatland change by several. The explanation for the connection was best described by L., and employee from an NGO, who explained that the mines create dark colored dust that settles on glaciers making the surface area dark and accelerating melting rates.

Peatland Uses and Benefits

Peatland uses and benefits can be split into divergent resource uses. *Arrieros* (donkey and mule drivers) and pastoralists view the peatlands primarily as a resource for their livestock, whereas tourists and hikers view them as possible camping spots and sources for clean water. Many pastoralists explained that the peatlands provide essential livestock fodder. They are places where non-native livestock spend most of their lives, as D. noted: "we raise animals on the bofedales. They are basic for our existence." A different D. said that "they are useful for our animals, like horses, donkeys, and cows." H. explained that "we use them for grazing. They are important for our animals. And for viscachas. They come down out of the rocks and eat." Two participants offered that that peatlands are good for producing *tocosh*, a traditional Andean immune-boosting product. Shortly after they are harvested, potatoes are placed in cold running water where they ferment over several months. The fermentation process produces antibiotic qualities and when consumed, *tocosh* has curative and immune-strengthening properties.

In comparison, tourists view the peatlands – especially the drier margins – as suitable camping spots because they are flat and generally free of rocks even though mosquitos can be a nuisance. In fact, the park designates camping areas near peatlands in Llanganuco, Quilcayhuanca, and Carhuascancha. Yet tourists recognize the biological and ecological value of peatlands. A., visiting from Argentina, commented that peatlands "are a pure space. They have a particular role in nature...it's water that is clean. It's a place to care for, that tourists need to care for and not pollute."

DISCUSSION

The quantitative results presented in Chapters 3, 4, and 5 demonstrate that peatlands in HNP are changing; they are decreasing in area and processes of fragmentation, shrinkage, attrition, and isolation are resulting in peatland patch shapes that are simpler and smaller. Loss of glacier area and decreasing streamflow are associated with decreasing peatland area, but in the future, precipitation may take over as a driving force of change. Heterogeneity from valley-to-valley is high and plant species richness and diversity are high. The complete flora for peatlands in HNP is probably higher than measured because of the high valley-to-valley heterogeneity. Elevation plays a role in vegetation structure and species richness. The most important abiotic factors contributing to plant community composition are elevation, CEC, % organic matter, and bulk density.

The ecological oral histories corroborate and validate these results and they go a step farther by providing richer context and a deeper temporal dimension. Participants report that they are seeing decreases in spatial extent and these observations are accompanied by other perceptions that wetlands are drying out and are now not as "pretty" as they once were. Participants confirm that negative ecological impacts are occurring, a biophysical effect that was inferred in Chapter 3 from fragmentation, shrinkage, attrition, and isolation detected by the pattern metrics. For example, the frogs have disappeared, the extent of *Plantago tubulosa* (*zampa*) is smaller, livestock are suffering, and trout are dying. These ecological impacts are linked to declines in water quantity and quality, similar to findings reported from southern Peru (Postigo 2014). The causes of these shifts are difficult to identify and there may be multiple causal factors working concurrently and in tandem (although widespread amphibian extinction has been linked to fungal infections (Wake and Vredenburg 2008)). In Quechua, C., a female pastoralist, eloquently captured all of these themes:

> "These days they (bofedales) aren't in such good condition, not like in the past. Now in the wetlands you even find *alicuya* (a livestock parasite) and they aren't as wet as before. I remember in my childhood the wetlands with crystalline water and the springs even clearer, but today you look at the wetlands and they are covered by a red layer that rises up from deep in the earth. It's like a red material that infects the wetlands in the same way that the *alicuya* does. Furthermore, you notice an oily layer on the surface of the water…In the past the wetlands were beautiful. They had plenty of water and even the animals fed in them. Now the animals don't like the grass because it stinks. The damage to the wetlands was caused by the earthquake of 1970."

The ecological oral histories reveal that peatlands are an important resource for pastoralists and *arrieros* because their animals use them for grazing, thus supporting their livelihoods and well-being. The relationship between users and peatlands explicitly connects the biophysical template to the social template via ecosystem services whereby the peatlands provide critical resource benefits to local people (See Figure 6.1). Based on the ecological oral histories, it becomes evident that livelihoods are affected by ecological change through shrinking peatlands that are drying out. Livestock are skinnier because pasture conditions are worsening and therefore the animals are worth less in the local market. The connection to the market is a critical point. Pastoralists use livestock to supplement incomes and bolster financial security; animals are sold when funds are needed (Lipton 2014). As a consequence, peatland loss and degradation is coupled with unhealthy and skinner livestock that translates to financial insecurity that then compromises livelihoods and well-being. None of these couplings were revealed in the previous chapters supporting the agrument that ecological oral histories give life and character to the Cartesian view from no-where (Haraway 1991; and as cited in Nightingale 2003) from the perspective of people who have experienced and observed peatland changes over their lifetimes. My research shows that when quantitative and qualitative methods are integrated as required by the PPD model and the CPG paradigm, a richer narrative of peatland change emerges than would be available from a single epistemology. Taken together, the three research questions that guide this dissertation are larger than any one academic discipline thus necessitating a design that bridges distinct tradtions to answer the research questions. Adopting and integrating multiple epistemologies produces new knowledge about a rapidly changing landscape that would otherwise not be accessible.

An important premise of this chapter is that the results of ecological oral histories are as equally valid as the quantitative results presented in previous chapters; however, there are some drawbacks that should be noted, although these are not unlike caveats associated with quantitiative methods (i.e. remote sensing accuracy assessments). Other researchers have recognized that oral histories may not always be accurate (Ritchie 2003; Trueman, Hobbs, and Van Niel 2013). Participants tend to recall what is important to them and this may not be what the researcher is inquiring about. In this case, discrepancy occurred not with pastoralists or *arrieros*, but with tourists who do not have in-depth temporal knowledge of the landscapes since they are visiting the region for short periods of a time and only occasionally. Additional interviews are ongoing and will continue to focus on participants who are older (> 55 years of age) because they have memories that reach further back in time. Relative to the land change analysis (Chapter 3) that spanned from 1987 to 2010, there is a temporal mismatch. The ecological oral histories date back to the participant's childhood, or early years. In some cases this could refer as far back as 40-60 years, or to the 1950s – 1970s, substantially longer than the 23-year trajectory. In effect, the ecological oral histories could reflect change that has occurred over a much longer temporal extent than the land change analysis. Archival sources and other written accounts – if they are available – could provide additional material to bridge this temporal mismatch. There is also an issue of spatial scale. The Chapter 3 analysis corresponds to the extent of the park, or the landscape level. In contrast, an interview participant is most likely reporting on observations that pertain to a single valley that their community oversees. *Comunidades Campesinas* (CCs, peasant communities) maintain land use rights within the park boundaries that include limited rights to graze livestock in designated valleys (Gilbert 2015). Pasture use groups communally designate zones within the valleys for grazing (Lipton 2014). A participant in the ecological oral histories is likely to have a "segmented view of the landscape" that can limit observations of patterns of change at the landscape scale (Jiang 2003, 229). Thus there is a spatial scalar mismatch between the remote sensing analysis that was performed at the landscape level and the ecological oral histories that correspond to the valley level. But because participants were interviewed from various CCs that use different valleys, it is reasonable to extrapolate from the valley level to create a plausible landscape level narrative.

Finally, through the process of integrating quantitative and qualitative results and presenting preliminary dissertation results to in-country partners, I have identified refinements to the PPD model generally and more specifically to the study area (Figure 6.1). Overall the model performs well, especially as a heuristic device used iteratively. Insight into the future of peatlands, to be discussed in the final section, emerged after integrating the quantitative and qualitative results. Only after identifying the ecosystem services associated with peatlands and the overall context of conservation and park-community relationships was I able to imagine possible human behaviors that could maintain and protect the peatlands. A refinement that I recommend is regarding disturbances. At the center of the initial figure adapted for HNP (Figure 1.2) are two disturbance types, presses and pulses, distinguished by chronicity and abruptness, respectively. In practice, determining the thresholds between presses and pulses is difficult and highly dependent upon the process, observer, and timescale. For example, glacier mass balance fluctuates over longer time scales (Chapter 2). From a human perspective, glacier recession is a press, but over geologic timescales, recession would be a pulse with major advances and retreats occurring on the scale of 100s and 1000s of years. Distinguishing between a press and a pulse is not necessarily useful and may only create confusion, thus I advocate using “disturbances”, a term that is widely accepted in ecology (Chapin, Kofinas, and Folke 2009). The ecological oral histories revealed that the ecosystem service of provisioning should include *tocosh* production. Culturally, peatlands were expected to have a spiritual value because landscape features in mountains can be associated with spirits or deities. In this case, no myths or legends were reported by participants, nor did they report making offerings. Differentiation of the social template into only human behavior and human outcomes is simplistic, especially because factors such as age, gender, and land tenure have no place in the model. Overly simplistic models were critiqued by Ostrom (2009), who developed a detailed SES framework focusing on sustainability. In addition, human behaviors and outcomes are unique for individuals, households, communities, and institutions. Practically speaking, the model is limited in terms of social scale and future refinements could improve the social template to better reflect the complexity of human systems on multiple scales.

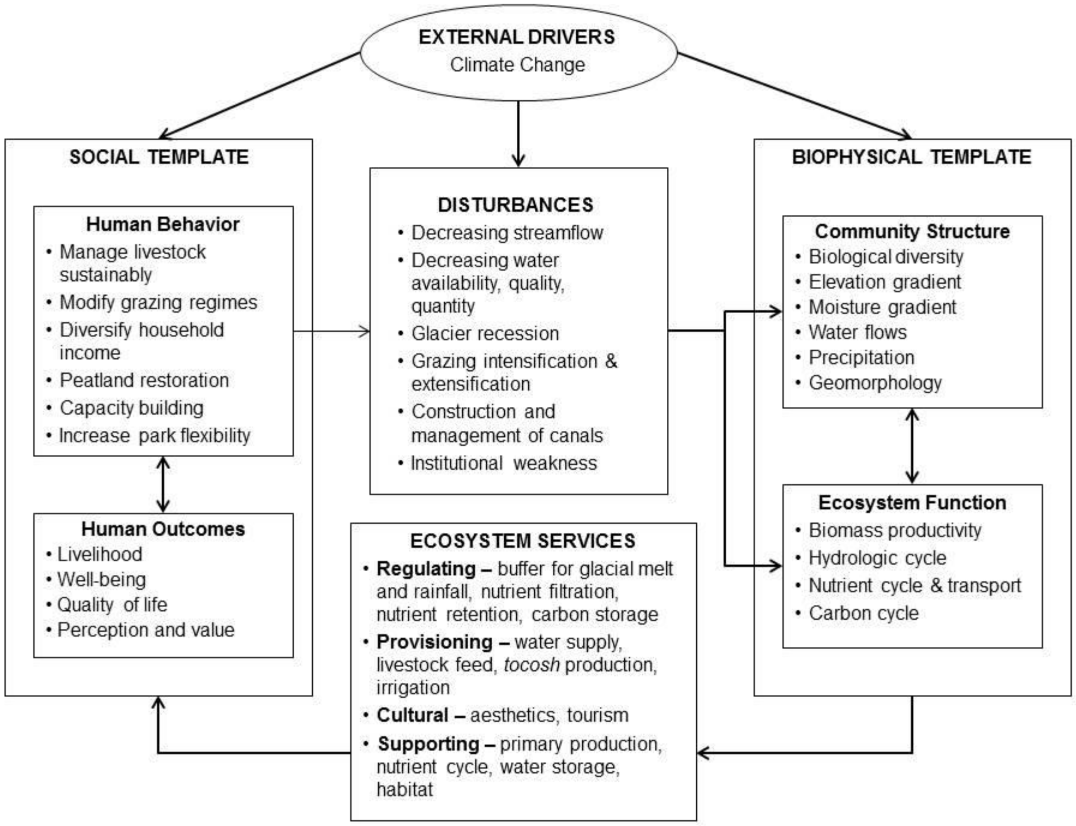


Figure 6.1. PPD framework revised for Huascarán National Park based on this dissertation.

CONCLUSION: THE FUTURE OF PEATLANDS IN HUASCARÁN NATIONAL PARK

With ample evidence that the peatlands are changing, what then does the future hold for these ecosystems? Looking again at the revised PPD model (Figure 6.1), what human behaviors affect peatlands? In other words, what decisions can be made by the peatland stewards – CCs, NGOs, state institutions – to manage these ecosystems in the future? NGOs promote themselves as well-intentioned, but suffer from sporadic funding and high staff turnover, among other serious issues that have problematized conservation efforts and hindered long-term success (Sundberg 1998). Park management, although comprised of exceptionally dedicated and passionate personnel, is limited by inadequate funding and top-down bureaucratic rigidity. Historic distrust between CCs and park governance dates back to a series of shocks that occurred over a short span as explained by Lipton (2014): the Agrarian Reform of 1969, the 1970 earthquake, and the establishment of the park in 1975. The legacy of distrust between the park, an agent of the federal government, and CCs, has its roots in the quick succession of these historic and deeply influential events.

With their close ties and deep knowledge of resource use, I agree with Lipton (2014) and Gilbert (2015) that the CCs should be the leading stewards of peatlands in the future. To date, the communities have not developed any management programs for peatlands in the park even though they know that their livelihoods are inextricably linked to the peatlands (at least this is true for Quilcayhuanca, Carhuascancha and Llanganuco). The reasons for not developing management programs were beyond the scope of this dissertation, but could include complex structural, political, historical, and cultural, explanations, among others; identifying and disentangling the reasons is a research opportunity. The CCs receive very little – if any – support from the park, with the exception of the community of Huashao near the mouth of the Llanganuco valley. As documented by Gilbert (2015), the community of Huashao and the park entered into a 5-year agreement beginning in the 2013 that granted exclusive rights to the community to provide goods and services to tourists entering the park at Llanganuco. In exchange for the economic benefits associated with the agreement, the community must dress in traditional attire to appeal to tourists, pay taxes to the park, provide two community members to work alongside park guards, implement a reforestry project, and promote the conservation and sustainable use of natural resources in the sector. This includes removing 250 cattle from inside the park boundaries (Gilbert 2015).

The case study from Huashao offers a glimpse into a possible form of future peatland stewardship when it is led by a coalition between CCs, park management, and NGOs. I conclude by recommending a series of actions that would simultaneously protect the ecological value and social benefits of peatlands in HNP. The foundation of these recommendations lies in assigning key conservation responsibilities to the CCs and allotting the responsibilities of capacity-building and sourcing technological solutions to the park and NGOs. Because community livelihoods and well-being are closely tied to livestock, it is unrealistic to expect communities to remove all livestock from inside the park boundaries. One solution would be for the park to work with the communities to develop grazing regimes that are designed to balance ecological integrity and livelihoods in the long-term. NGOs could help the park and communities identify grazing protocols from other high elevation grassland-peatland ecosystems that may have been effective. Together the park and NGO could facilitate dialogue and technology exchanges between the communities and other groups who have implemented effective grazing protocols. Herd compositions differ from community to community so there will be a need to develop grazing regimes depending on each community's herds. For example, cattle and horses dominate valleys on the west side of the park, whereas sheep dominate on the east side; grazing regimes should be customized for these community differences that would allow peatlands to maintain ecological integrity and restore them where possible. In valleys where herd sizes are very large, a portion of animals could be purchased similar to the purchase of 250 cattle in Llanganuco mentioned above.

In areas where peatlands have been heavily disturbed, restoration projects and efforts to reduce herd size should be attempted. In the last 9 months, The Mountain Institute, an NGO, has initiated a project with peatland restoration experts from Michigan Technological University and funded by the U.S. National Forest Service and USAID. The goals of the project are to help the park strengthen peatland management, improve scientific knowledge of mountain peatlands, and install pilot restoration projects in two valleys, Pucavado and Ulta. They have installed structures in several small degraded peatlands near roads to re-initiate hydrologic connectivity that will lead to ecosystem restoration. The Mountain Institute recognizes that local users are critical to the project's success and are working with CCs and the park to protect and restore peatlands (The Mountain Institute 2015). A project like this should also address construction and maintenance of canals that are dug by communities and maintained annually to dry out the peatlands and improve fodder for livestock. New canal construction should be prohibited and maintenance should be abandoned, allowing the canals to naturally fill in (canals that provide drinking water and supply irrigation should be exempt). To scale up and create a positive impact across the park, the project could install additional restoration projects and work closely with communities to communicate the value of restoration. Otherwise, communities are likely to damage or remove installations, as was observed in 2014 in Quilcayhuanca when communities destroyed newly installed exclosure plots and blamed damage on angry bulls. In the region, NGOs and other volunteer organizations have failed to develop successful projects and have left behind incomplete and abandoned efforts that are of no benefit to the communities and engender distrust against NGOs (Gilbert 2015). One of the grand challenges for the USFS-USAID funded restoration project will be to avoid the failed efforts of past projects and develop a long-lasting campaign that is valuable to communities and the park.

The future of peatlands in HNP will be determined by disturbances such as climate induced glacier recession, decreasing streamflow, and other shifting hydrologic inputs along with grazing pressure, canals, and institutional weakness. In practice, communities are the primary land use decision makers and can lead efforts to protect and maintain peatlands in the face of ongoing ecological change. These recommendations are centered on community-based initiatives to overcome institutional weakness that characterizes the national park. In a study that has been sometimes overwhelmingly negative in terms of the long-term ecological outlook for a tropical mountain ecosystem, the case study from Huashao offers a positive future. It demonstrates that mutually beneficial programs are feasible particularly when park management is given the flexibility to devise innovative strategies that delegate conservation responsibilities to communities.

Appendices

Appendix 1

Vascular Plant Taxa Found In Quadrats.

Num. of Quads represents the number of quadrats in which the species was observed. Endemism, Growth Form, Habitat (where available), and Known Geographic Distribution were obtained from Missouri Botanical Garden's Tropicos database (www.tropicos.org) and from Brako and Zarucchi (1993). † Indicates non-native/exotic species.

Taxa	Num. of Quads	Species Code	Endemic to Peru	Growth Form	Habitat	Known Geographic Distribution
APIACEAE						
Chaerophyllum andicola (Kunth) K.F. Chung	3	chaand	N	Herb	Grasslands, rocky slopes, shrublands	Ecuador, Peru
Daucus montanus Humb. & Bonpl. ex Spreng.	5	daumon	N	Herb	Disturbed areas, lomas, rocky slopes	Argentina, Bolivia, Chile, Colombia, Costa Rica, Ecuador, El Salvador, Guatemala, Honduras, Mexico, Peru, Venezuela
Eryngium humile Cav.	1	eryhum	N	Herb	Grasslands, rocky slopes	Colombia, Costa Rica, Ecuador, Panama, Peru, Venezuela
Lilaeopsis macloviana (Gand.) A.W. Hill	2	lilmac	N	Herb	Seasonally inundated areas, submerged, terra firme forests	Argentina, Bolivia, Chile, Ecuador, Peru

ASTERACEAE

Aphanactis villosa S.F. Blake	2	aphvil	N	Herb	Grasslands, rocky slopes, shrublands	Ecuador, Peru
Belloa piptolepis (Wedd.) Cabrera	3	belpip	N	Herb	Grasslands, shrublands	Argentina, Bolivia, Chile, Colombia, Ecuador, Peru, Venezuela
Bidens andicola Kunth	1	bidand	N	Herb	n.d.	Argentina, Bolivia, Colombia, Ecuador, Peru, Venezuela
Cotula mexicana (DC.) Cabrera	22	cotmex	N	Herb	Disturbed areas	Argentina, Bolivia, Chile, Colombia, Costa Rica, Ecuador, Mexico, Panama, Peru, Venezuela
Erigeron lanceolatus Wedd.	5	erilan	N	Herb	n.d.	Argentina, Bolivia, Peru
Gamochaeta cabrerae Anderb.	1	gamore	Y	Herb	n.d.	Peru
Hypochaeris meyeniana (Walp.) Benth. & Hook. f. ex Griseb	3	hypmey	N	Herb	n.d.	Bolivia, Ecuador, Peru
Hypochaeris taraxacoides Ball	6	hyptar	N	Herb	n.d.	Argentina, Bolivia, Chile, Peru
Loricaria sp.	1	lorsp1	N	Herb	n.d.	n.d.
Lucilia kunthiana (DC.) Zardini	15	luckun	N	Herb	Grasslands, shrublands	Colombia, Ecuador, Peru, Venezuela
Oritrophium limnophilum (Sch. Bip.) Cuatrec.	34	orilim	N	Herb	n.d.	Bolivia, Colombia, Ecuador, Peru, Venezuela
Oritrophium sp.	1	orisp1	N	Herb	n.d.	n.d.
Paranephelius ovatus A. Gray ex Wedd.	8	parova	Y	Herb	n.d.	Bolivia, Peru

Senecio breviscapus DC.	2	senbre	N	Herb	n.d.	Argentina, Bolivia, Chile, Peru, South Africa
Senecio condimentarius Cabrera	1	sencon	N	Herb	n.d.	Peru
Senecio repens var. macbridei (Cuatrec.) Cabrera	3	senrep	N	Herb	n.d.	Peru
Werneria nubigena Kunth	23	wernub	N	Herb	Disturbed areas	Bolivia, Colombia, Costa Rica, Ecuador, Guatemala, Mexico, Panama, Peru
Werneria pygmaea Gillies ex Hook. & Arn.	20	werpyg	N	Herb	n.d.	Argentina, Bolivia, Chile, Colombia, Ecuador, Peru, Venezuela
Werneria sp.	1	wersp1	N	Herb	n.d.	n.d.
BRASSICACEAE						
Lepidium meyenii Walp.	2	lepmey	N	Herb	Disturbed areas	Bolivia, Chile, Peru,
CAMPANULACEAE						
Hypsela reniformis (Kunth) C. Presl	24	hypren	N	Herb	n.d.	Bolivia, Chile, Colombia, Ecuador, Peru
CARYOPHYLLACEAE						
Arenaria serpens Kunth	1	arepal	N	Herb	n.d.	Argentina, Bolivia, Chile, Ecuador, Mexico, Peru
Cerastium mucronatum Wedd.	1	cermuc	N	Herb	n.d.	Bolivia, Peru
Cerastium sp.	1	cersp1	N	Herb	n.d.	n.d.
Drymaria engleriana (Muschl.) Baehni & J.F. Macbr.	1	dryeng	Y	Herb	n.d.	Peru

CYPERACEAE

Carex bonplandii Kunth	27	carbon	N	Herb	Disturbed areas, grasslands, riversides	Argentina, Belize, Bolivia, Brazil, Colombia, Costa Rica, Ecuador, El Salvador, Guatemala, Honduras, Mexico, Nicaragua, Panama, Peru, U.S.A., Venezuela
Carex hebetata Boott	3	carheb	Y	Herb	Grasslands	Peru
Carex pygmaea Boeckeler	19	carpyg	N	Herb	n.d.	Bolivia, Colombia, Costa Rica, Ecuador, Panama, Peru
Eleocharis albibracteata Nees & Meyen ex Kunth	42	elealb	N	Herb	Bogs, disturbed areas, grasslands	Argentina, Chile, Ecuador, Guatemala, Peru,
Isolepis cf. cernua	2	isocer	N	Herb	n.d.	n.d.
Luzula racemosa Desv.	3	luzrac	N	Herb	Grasslands, rocky slopes	Argentina, Bolivia, Chile, Colombia, Costa Rica, Ecuador, Guatemala, Mexico, Peru, Venezuela
Oreobolopsis sp.	7	oresp1	N	Herb	n.d.	n.d.
Oreobolus obtusangulus Gaudich.	17	oreobt	N	Herb	Bogs, disturbed areas, grasslands, riversides	Argentina, Bolivia, Chile, Colombia, Ecuador, Peru, Venezuela
Phylloscirpus acaulis (Phil.) Geotgh. & D.A. Simpson	12	phyaca	N	Herb	Disturbed areas	Argentina, Bolivia, Chile, Ecuador, Peru
Phylloscirpus deserticola (Phil.) Dhooge & Goetgh.	15	phydes	N	Herb	n.d.	Argentina, Bolivia, Chile, Ecuador, Peru
Rynchospora sp.	1	rynsp1	N	Herb	n.d.	n.d.

Trichophorum rigidum (Boeckeler) Goetgh., Muasya & D.A. Simpson	4	tririg	N	Herb	n.d.	Argentina, Bolivia, Brazil, Chile, Colombia, Costa Rica, Ecuador, Peru
EQUISETACEAE						
Equisetum bogotense Kunth	1	equbog	N	Herb	n.d.	Argentina, Bolivia, Chile, Ecuador, Panama, Paraguay, Peru, Uruguay, Venezuela
ERICACEAE						
Pernettya prostrata(Cav.) DC.	14	perpro	N	Shrub	Disturbed areas, forests, grasslands, rocky slopes	Argentina, Bolivia, Colombia, Costa Rica, Ecuador, Guatemala, Honduras, Mexico, Nicaragua, Panama, Peru, Venezuela
FABACEAE						
Astragalus garbancillo Cav.	1	astgar	N	Herb/ Shrub	Grasslands, shrublands	Argentina, Bolivia, Peru
Lupinus paniculatus Desr.	1	luppan	N	Herb/ Subshrub	Riversides, rocky slopes	Bolivia, Colombia, Ecuador, Peru, Venezuela
Trifolium amabile Kunth	8	triama	N	Herb	n.d.	Argentina, Bolivia, Chile, Colombia, Costa Rica, Ecuador, Guatemala, Mexico, Peru, U.S.A.
GENTIANACEAE						
Gentiana sedifolia Kunth	37	gensed	N	Herb	Grasslands, rocky slopes	Argentina, Bolivia, Chile, Colombia, Costa Rica, Ecuador, Guatemala, Panama, Peru, Venezuela
Gentianella bicolor Wedd.	1	genbic	N	Herb	Grasslands	Peru
Gentianella poculifera (Gilg) Zarucchi	1	genpoc	Y	Herb	Grasslands	Peru

Gentianella tristicha Gilg	9	gentri	Y	Herb	Grasslands, shrublands	Peru
Halenia caespitosa Gilg	12	halcae	N	Herb	Grasslands, riversides, shrublands	Bolivia, Peru
Halenia umbellata (Ruiz & Pav.) Gilg	2	halumb	N	Herb	Bogs, grasslands, riversides, rocky slopes	Bolivia, Peru
HYPERICACEAE						
Hypericum laricifolium Juss.	3	hyplar	N	Shrub, subshrub, tree	Cloud forests, shrublands	Bolivia, Colombia, Ecuador, Peru, Venezuela
Hypericum silenoides Juss.	2	hypsil	N	Herb	Rocky slopes	Argentina, Bolivia, Chile, Colombia, Ecuador, Mexico, Peru
IRIDACEAE						
Sisyrinchium caespitificum Kraenzl.	1	siscae	N	Herb	Grasslands	Ecuador, Peru
Sisyrinchium chilense Hook.	1	sischi	N	Herb	Grasslands	Argentina, Bolivia, Chile, Ecuador, Peru
Sisyrinchium trinerve Baker	5	sistri	N	Herb	Grasslands	Bolivia, Chile, Colombia, Costa Rica, Ecuador, Peru, Venezuela
JUNCACEAE						
Distichia muscoides Nees & Meyen	1	dismus	N	Herb	Grasslands	Argentina, Bolivia, Chile, Colombia, Ecuador, Peru,
Juncus arcticus Willd.	1	junart	N	Herb	Grasslands, riversides	Argentina, Bolivia, Canada, Chile, Colombia, Ecuador, Guatemala, Mexico, Norway, Peru, United States

Juncus ebracteatus E. Mey.	39	junebr	N	Herb	Rocky slopes, shrublands	Bolivia, Guatemala, Mexico, Peru
LAMIACEAE						
Stachys arvensis L. †	1	staarv	N	Herb	Disturbed areas, rocky slopes	Argentina, Bolivia, Brazil, China, Ecuador, Russia, South Africa, United States, Venezuela
LYCOPODIACEAE						
Huperzia crassa (Humb. & Bonpl. ex Willd.) Rothm.	21	hupcra	N	Herb	n.d.	Bolivia, Caribbean, Colombia, Costa Rica, Ecuador, Guatemala, Mexico, Panama, Paraguay, Peru, Venezuela
MARCHANTIACEAE						
Marchantia polymorpha L.	2	marpol	N	Herb	n.d.	Nonvascular. Brazil, Chile, China, Japan, Peru Russia, United States.
MONTIACEAE						
Calandrinia acaulis Kunth	1	calaca	N	Herb	Rocky slopes	Argentina, Bolivia, Chile, Colombia, Costa Rica, Ecuador, Guatemala, Mexico, Peru, Venezuela
ONAGRACEAE						
Epilobium pedicellare C. Presl	1	epiped	N	Herb	Riversides	Bolivia, Chile, Peru
Epilobium sp.	2	episp1	N	Herb	n.d.	n.d.

Oenothera multicaulis Ruiz & Pav.	2	oenmul	N	Herb	Grasslands, shrublands	Bolivia, Costa Rica, Ecuador, Guatemala, Mexico, Peru, Venezuela
OPHIOGLOSSACEAE						
Ophioglossum crotalophoroides Walter	1	ophcro	N	Herb	n.d.	Argentina, Bolivia, Brazil, Caribbean. Chile, Colombia, Costa Rica, Ecuador, Guatemala, Honduras, Mexico, Nicaragua, Peru, United States, Uruguay, Venezuela
ORCHIDACEAE						
Aa sp.	1	aaspe1	N	Herb	n.d.	n.d.
Myrosmodes paludosum (Rchb. f.) P. Ortiz	1	myrpal	N	Herb	Grasslands, shrublands	Bolivia, Colombia, Peru, Venezuela
Myrosmodes pumilio (Schltr.) C. Vargas	18	myrpum	N	Herb	n.d.	Bolivia, Peru
OROBANCHACEAE						
Bartsia diffusa Benth.	2	bardif	N	Herb	Grasslands	Bolivia, Peru
Bartsia melampyroides (Kunth) Benth.	3	barmel	N	Herb	Disturbed areas, grasslands	Bolivia, Ecuador, Peru
Castilleja pumila (Benth.) Wedd.	11	caspum	N	Herb	Grasslands	Bolivia, Chile, Colombia, Ecuador, Peru
Castilleja virgatoides Edwin	5	casvir	Y	Herb	n.d.	Peru
PHRYMACEAE						
Mimulus glabratus Kunth	2	mimgla	N	Herb	Riversides	Argentina, Bolivia, Canada, Chile, Colombia, Ecuador, Guatemala, Mexico, Nicaragua, Peru, United States, Venezuela

PLANTAGINACEAE						
Plantago lamprophylla Pilg.	1	plalam	N	Herb	Disturbed areas, grasslands, rocky slopes, shrublands	Bolivia, Peru
Plantago tubulosa Decne.	43	platub	N	Herb	Bogs, grasslands, rocky slopes	Argentina, Bolivia, Chile, Colombia, Ecuador, Guatemala, Mexico, Peru
POACEAE						
Aciachne pulvinata Benth.	1	acipul	N	Herb	Bogs, grasslands, rocky slopes	Bolivia, Costa Rica, Ecuador, Peru, Venezuela
Agrostis breviculmis Hitchc.	17	agrbre	N	Herb	Disturbed areas, grasslands	Argentina, Bolivia, Brazil, Canada, Chile, Colombia, Ecuador, Peru, Venezuela
Agrostis tolucensis Kunth	10	agrtol	N	Herb	Disturbed areas, grasslands	Argentina, Bolivia, Chile, Colombia, Costa Rica, Ecuador, Guatemala, Mexico, Panama, Peru, Venezuela
Calamagrostis brevifolia (J. Presl) Steud.	12	calbre	N	Herb	Disturbed areas, grasslands, rocky slopes	Argentina, Bolivia, Peru
Calamagrostis eminens (J. Presl) Steud.	2	calemi	N	Herb	Bogs, riversides	Argentina, Bolivia, Chile, Colombia, Peru
Calamagrostis heterophylla (Wedd.) Pilg.	3	calhet	N	Herb	Grasslands, riversides	Argentina, Bolivia, Chile, Ecuador, Peru, Venezuela

Calamagrostis recta (Kunth) Trin. ex Steud.	3	calrec	N	Herb	Grasslands, riversides, shrublands	Argentina, Bolivia, Colombia, Ecuador, Peru, Venezuela
Calamagrostis rigescens (J. Presl) Scribn.	36	calrig	N	Herb	Grasslands, riversides, swamps	Argentina, Bolivia, Chile, Ecuador, Mexico, Peru
Calamagrostis tarmensis Pilg.	1	caltar	N	Herb	Grasslands, rocky slopes	Argentina, Bolivia, Ecuador, Peru
Calamagrostis vicunarum (Wedd.) Pilg.	27	calvic	N	Herb	Grasslands, rocky slopes	Argentina, Bolivia, Chile, Ecuador
Eragrostis peruviana (Jacq.) Trin.	2	poaper	N	Herb	Disturbed areas, lomas	Peru, Chile
Festuca nigriflora (Hitchc.) Negritto & Anton	1	fesnig	N	Herb	n.d.	Peru
Festuca procera Kunth	3	fespro	N	Herb	Disturbed areas	Bolivia, Colombia, Ecuador, Peru
Muhlenbergia ligularis (Hack.) Hitchc.	22	muhlig	N	Herb	Disturbed areas, grasslands	Argentina, Bolivia, Colombia, Costa Rica, Ecuador, Guatemala, Peru, Venezuela
Muhlenbergia peruviana (P. Beauv.) Steud.	1	muhper	N	Herb	Forests, grasslands	Argentina, Bolivia, Chile, Ecuador, Guatemala, Mexico, Peru, United States
Nassella brachyphylla (Hitch.) Barkworth	1	nasbra	N	Herb	Grasslands	Argentina, Bolivia, Ecuador, Peru
Nassella inconspicua (J. Presl) Barkworth	3	nasinc	N	Herb	Grasslands	Argentina, Bolivia, Colombia, Ecuador, Peru

Pennisetum clandestinum Hochst. ex Chiov. †	1	pencla	N	Herb	Disturbed areas, riversides	Argentina, Bolivia, Brazil, Burundi, Caribbean, China, Colombia, Costa Rica, Ecuador, Ethiopia, Greece, Guatemala, Honduras, India, Kenya, Mexico, Nicaragua, Panama, Paraguay, Peru, Rwanda, South Africa, Tanzania, Uganda, United States, Uruguay, Venezuela, Congo
Poa annua L.	3	poaann	N	Herb	Disturbed areas, forests, grasslands, lomas, riversides	Global
Poa serpaiana Refulio	2	poaser	N	Herb	n.d.	Argentina, Bolivia, Chile, Peru
POLYGONACEAE						
Muehlenbeckia volcanica (Benth.) Endl.	1	muevol	N	Herb, shrub, vine	Disturbed areas, elfin forests, grasslands, riversides, rocky slopes, shrublands	Bolivia, Brazil, Colombia, Costa Rica, Ecuador, Guatemala, Mexico, Peru, Venezuela

Rumex acetosella L.[†]	1	rumace	N	Herb	Disturbed areas, rocky slopes, shrublands	Argentina, Bolivia, Brazil, Canada, Caribbean, Chile, China, Colombia, Costa Rica, Ecuador, Greenland, Guatemala, India, Japan, Kazakhstan, Mexico, Mongolia, New Zealand, Panama, Peru, Russia, South Africa, South Korea, United States, Venezuela
RANUCULACEAE						
Ranunculus flagelliformis Sm.	2	ranfla	N	Herb	Aquatic	Argentina, Bolivia, Brazil, Caribbean, Chile, Colombia, Costa Rica, Ecuador, Guatemala, Mexico, Paraguay, Peru, Uruguay, Venezuela
Ranunculus limoselloides Turcz.	1	ranlim	N	Herb	Aquatic	Bolivia, Colombia, Ecuador, Peru, Venezuela
Ranunculus praemorsus Humb., Bonpl. & Kunth ex DC.	1	ranpra	N	Herb	Disturbed areas, grasslands	Argentina, Bolivia, Chile, Colombia, Costa Rica, Ecuador, Mexico, Panama, Peru, Venezuela
ROSACEAE						
Lachemilla aphanoides (Mutis ex L. f.) Rothm.	1	lacaph	N	Herb	Bogs, disturbed areas, forests	Bolivia, Colombia, Costa Rica, Ecuador, Guatemala, Mexico, Panama, Peru, Venezuela

Lachemilla orbiculata (Ruiz & Pav.) Rydb.	7	lacorb	N	Herb	Bogs, cloud forests, disturbed areas, grasslands, shrublands	Bolivia, Colombia, Ecuador, Peru, Venezuela
Lachemilla pinnata (Ruiz & Pav.) Rothm.	35	lacpin	N	Herb	Cloud forests, disturbed areas, grasslands, shrublands	Argentina, Bolivia, Colombia, Chile, Costa Rica, Ecuador, Guatemala, Mexico, Peru, Venezuela
Lachemilla sp.	1	lacsp1	N	n.d.	n.d.	n.d.
Lachemilla vulcanica (Schltdl. & Cham.) Rydb.	1	lacvul	N	Herb	Grasslands, rocky slopes, shrublands	Bolivia, Colombia, Ecuador, El Salvador, Guatemala, Mexico, Peru, Venezuela
RUBIACEAE						
Galium corymbosum Ruiz & Pav.	3	galcor	N	Herb	Disturbed areas, grasslands, riversides, rocky slopes, shrublands	Bolivia, Colombia, Ecuador, Peru, Venezuela

Nertera granadensis (Mutis ex L. f.) Druce	2	nergra	N	Herb	Disturbed areas, forests, grasslands	Argentina, Australia, Bolivia, Caribbean, Chile, China, Colombia, Costa Rica, Ecuador, El Salvador, Guatemala, Hawaii, Honduras, Indonesia, Malaysia, Mexico, New Guinea, New Zealand, Nicaragua, Panama, Philippines, Taiwan, Venezuela
SCROPHULARIACEAE						
Limosella subulata E. Ives	10	limsub	N	Herb	Submerged	Canada, Ecuador, Peru, United States, Venezuela
VALERIANACEAE						
Belonanthus sp.	1	belsp1	N	n.d.	n.d.	n.d.

APPENDIX 2

Confusion matrices for classification accuracy assessments.
Reference data are in columns and land cover classes are in rows.

1999

	Barren	**Puna**	**Wetland**	**Snow/Ice**	**Water**	**Shadow**	**Row Total**
Barren	84	1	1	11	0	3	100
Puna	4	93	1	0	1	1	100
Wetland	1	15	82	0	0	2	100
Snow/Ice	0	0	0	98	0	2	100
Water	2	0	0	0	98	0	100
Shadow	3	2	0	1	0	94	100
Column Total	94	111	84	110	99	102	
Diagonal Total	549						

2010

	Barren	**Puna**	**Wetland**	**Snow/Ice**	**Water**	**Shadow**	**Row Total**
Barren	63	2	0	5	0	8	78
Puna	10	58	0	0	5	4	77
Wetland	1	13	68	0	0	3	85
Snow/Ice	0	0	0	66	0	0	66
Water	1	2	0	0	74	0	77
Shadow	2	25	17	6	0	36	86
Column Total	77	100	85	77	79	51	
Diagonal Total	365						

APPENDIX 3

Landscape Ecology metrics for all classes through time.
See Chapter 3 for explanations of metrics.

Number of Patches

	1987	1990	1995	1999	2005	2010
Barren	5503	5443	6222	5729	9202	6473
Puna	2515	4092	3054	2796	5669	2895
Wetland	6912	8768	8626	8870	8667	8399
Snow/Ice	1427	397	655	548	324	366
Water	294	295	284	406	402	436
Shadow	3455	4806	3649	2918	3157	8273
Cloud	790	0	0	0	366	0

Total Edge (kilometers)

	1987	1990	1995	1999	2005	2010
Barren	11,676	17,876	17,616	16,045	18,712	16,547
Puna	15,065	13,877	11,821	13,029	21,583	12,141
Wetland	4,377	5,181	4,323	2,682	4,741	9,889
Snow/Ice	3,709	5,793	5,151	5,394	5,159	3,501
Water	365	2,315	3,101	2,837	2,441	2,572
Shadow	5,178	363	368	426	435	453
Cloud	755	0	0	0	347	0

Mean Patch Area (hectares)

	1987	1990	1995	1999	2005	2010
Barren	9.69	16.65	10.35	14.37	10.34	9.85
Puna	71.67	37.00	61.45	63.40	27.64	66.96
Wetland	1.79	1.59	1.61	1.70	1.63	0.81
Snow/Ice	48.29	151.05	85.46	101.05	152.90	125.25
Water	8.24	8.12	8.78	6.54	6.94	6.55
Shadow	6.19	4.61	4.48	2.71	6.74	3.32
Cloud	2.34	0.00	0.00	0.00	2.51	0.00

Mean Core Area (10 m buffer, hectares)

	1987	1990	1995	1999	2005	2010
Barren	9.69	16.65	10.35	14.37	10.34	9.85
Puna	71.67	37.00	61.45	63.40	27.64	66.96
Wetland	1.79	1.59	1.61	1.70	1.63	0.81
Snow/Ice	48.29	151.05	85.46	101.05	152.90	125.25
Water	8.24	8.12	8.78	6.54	6.94	6.55
Shadow	6.19	4.61	4.48	2.71	6.74	3.32
Cloud	2.34	0.00	0.00	0.00	2.51	0.00

Mean Core Area (50 m buffer, hectares)

	1987	1990	1995	1999	2005	2010
Barren	4.74	10.41	5.81	8.86	5.73	5.44
Puna	54.21	25.09	44.82	46.85	17.52	50.76
Wetland	0.50	0.42	0.42	0.48	0.45	0.12
Snow/Ice	41.86	134.50	72.50	86.50	131.39	104.91
Water	4.99	4.90	5.37	3.87	4.16	3.91
Shadow	2.69	1.90	1.72	0.73	3.15	0.80
Cloud	0.58	0.00	0.00	0.00	0.81	0.00

Shape Index (unitless)

	1987	1990	1995	1999	2005	2010
Barren	1.68	1.56	1.56	1.54	1.66	1.57
Puna	1.42	1.54	1.41	1.40	1.57	1.43
Wetland	1.26	1.25	1.25	1.25	1.25	1.21
Snow/Ice	1.59	1.53	1.60	1.53	1.64	1.60
Water	1.26	1.25	1.26	1.23	1.23	1.23
Shadow	1.62	1.55	1.50	1.47	1.58	1.60
Cloud	1.63	0.00	0.00	0.00	1.65	0.00

Cohesion (unitless)

	1987	1990	1995	1999	2005	2010
Barren	98.09	99.68	98.68	99.48	99.10	98.92
Puna	99.91	99.87	99.94	99.92	99.81	99.91
Wetland	92.08	92.21	91.61	92.81	92.31	84.07
Snow/Ice	99.55	99.57	99.52	99.61	99.48	99.46
Water	92.35	92.26	92.63	92.03	92.26	92.09
Shadow	94.36	93.05	92.41	88.78	94.90	92.24
Cloud	86.08	0.00	0.00	0.00	88.98	0.00

Euclidean Nearest Neighbor (meters)

	1987	1990	1995	1999	2005	2010
Barren	141.06	136.17	141.50	135.08	103.52	134.91
Puna	93.00	98.06	91.09	88.92	92.98	86.20
Wetland	130.93	126.78	129.59	125.57	125.92	132.84
Snow/Ice	162.41	194.21	174.67	151.03	164.62	156.31
Water	919.37	912.53	949.72	782.73	790.35	831.02
Shadow	177.03	167.44	173.80	203.72	194.35	136.18
Cloud	308.48	0.00	0.00	0.00	280.52	0.00

Appendix 4

Users and Producers Accuracy and Conditional Kappa for Classified 1987 and 1995 images.

	1987			1995		
Class	**Producers**	**Users**	**Conditional Kappa**	**Producers**	**Users**	**Conditional Kappa**
Barren	81.15%	85.34%	82.71%	79.17%	80.67%	76.43%
Puna	81.46%	92.36%	90.17%	75.72%	83.44%	78.00%
Wetland	94.62%	87.13%	85.44%	88.64%	75.73%	72.24%
Snow/Ice	97.37%	92.50%	91.25%	90.98%	94.87%	93.79%
Lake	95.19%	99.00%	98.85%	97.94%	95.00%	94.20%
Shadow	91.76%	73.58%	70.44%	93.55%	83.65%	81.15%
Cloud	91.35%	95%	94.25%	N/A	N/A	N/A

APPENDIX 5

Soil Analysis Results provided by the Laboratorio de Suelos, Universidad Nacional Agraria La Molina.

Lab ID	Field ID	pH (1:1)	%OM	P (ppm)	K (ppm)	Sand	Loam	Clay	Textural Class	CEC	Bulk Density (g/cm3)
10776	LP1, 10 Cm.	4.76	1.82	0.1	30	58	40	2	Fr.A.	3.20	1.04
10777	LP3, 10 Cm.	4.23	10.05	4.8	91	78	20	2	A.Fr.	11.68	0.47
10779	LP5, 10 Cm.	5.20	64.73	53.8	360	Suelo Orgánico				46.00	0.10
10781	LP7, 10 Cm.	4.96	5.51	0.5	52	50	44	6	Fr.A.	8.80	0.78
10782	LP11, 10 Cm.	4.19	7.11	4.8	63	54	44	2	Fr.A.	5.60	0.93
10783	LP13, 10 Cm.	4.54	9.35	3.2	74	68	30	2	Fr.A.	9.12	0.53
10784	LP15, 10 Cm.	4.19	57.63	168.5	776	Suelo Orgánico				49.60	0.10
10785	LP17, 10 Cm.	4.46	2.65	0.5	73	70	26	4	Fr.A.	4.80	1.17
10786	LP19, 10 Cm.	4.58	55.71	77.4	636	Suelo Orgánico				48.00	0.10
10787	LP21, 10 Cm.	4.67	67.73	80.9	1053	Suelo Orgánico				60.80	0.11
10789	LP23, 10 Cm.	4.69	3.64	8.5	37	90	8	2	A.	3.84	0.95
10790	LP25, 10 Cm.	4.67	19.44	6.7	233	70	22	8	Fr.A.	20.80	0.39
10792	LP29, 10 Cm.	4.49	6.08	3.3	134	70	26	4	Fr.A.	11.20	0.77
10793	LP31, 10 Cm.	4.89	4.22	0.2	93	40	52	8	Fr.L.	6.40	1.15
12583	QP 1	3.92	42.27	13.5	253	Suelo Orgánico				48.00	0.20
12584	QP 5	5.36	57.64	11.2	313	Suelo Organico				76.00	0.12
12585	QP 8	3.66	54.23	17.2	141	Suelo Organico				70.40	0.14
12586	QP 9	4.21	39.11	8.0	104	Suelo Organico				66.80	0.08
12587	QP 13	4.25	54.34	14.3	499	Suelo Organico				62.00	0.19
12588	QP 15	4.53	63.40	67.8	455	Suelo Organico				61.20	0.08
12589	QP 17	4.26	36.64	14.3	55	Suelo Organico				11.20	1.14
12590	QP 19	4.55	40.48	13.0	181	Suelo Organico				54.80	0.20
12591	QP 20	3.97	55.17	13.7	379	Suelo Organico				66.00	0.07
8579	QP 21	4.18	8.28	9.2	248	67	28	5	Fr.A.	10.88	ND

12592	QP 23	4.07	63.95	11.0	308	Suelo Organico				73.60	0.11
12593	QP 25	3.84	59.83	12.0	270	Suelo Organico				70.00	0.07
12594	QP 27	3.92	51.33	13.0	223	Suelo Organico				72.00	0.12
12595	QP 29	4.11	68.34	260.0	880	Suelo Orgánico				66.80	0.08
12596	QP 31	3.93	55.44	13.9	146	Suelo Orgánico				64.80	0.09
13511	CP2	4.67	61.48	31.5	353	Suelo Orgánico				51.20	0.27
13512	CP12	4.02	73.56	119.1	528	Suelo Orgánico				80.00	0.16
13513	CP16	4.66	2.45	3.6	26	61	36	3	Fr.A.	9.12	1.19
13514	CP18	4.02	73.01	80.9	351	Suelo Orgánico				32.32	0.11
13515	CP24	4.62	44.90	8.4	191	Suelo Orgánico				42.40	0.41
13516	CP26	4.73	81.52	103.0	1045	Suelo Orgánico				54.80	0.08
13517	CP28	4.86	71.36	58.3	416	Suelo Orgánico				60.00	0.07
13518	CP30	5.21	55.72	14.5	610	Suelo Orgánico				50.40	0.25
13519	CP34	4.58	64.77	108.6	591	Suelo Orgánico				54.40	0.10

APPENDIX 6

University of Texas Institutional Review Board Approval

OFFICE OF RESEARCH SUPPORT

THE UNIVERSITY OF TEXAS AT AUSTIN

P.O. Box 7426, Austin, Texas 78713 · Mail Code A3200
(512) 471-8871 · FAX (512) 471-8873

FWA # 00002030

Date: 06/19/15

PI: Kenneth R Young

Dept: Geography and the Environment

Title: Impacts of Glacier Recession on Mountain Peatlands

Re: IRB Exempt Continuing Review Determination for Protocol Number 2012-01-0009

Dear Kenneth R Young:

Your research was reviewed to determine if it still meets the requirements of research that is exempt from IRB review. It has been determined that it continues to meet the requirements.
Updated Qualifying Period: 06/19/2015 to 06/18/2018 . *Expires 12 a.m. [midnight] of this date.*
A continuing review report must be submitted in three years if the research is ongoing.

Responsibilities of the Principal Investigator:

Research that is determined to be Exempt from Institutional Review Board (IRB) review is not exempt from ensuring protection of human subjects. The following criteria to protect human subjects must be met. The Principal Investigator (PI):

1. Assures that all investigators and co-principal investigators are trained in the ethical principles, relevant federal regulations, and institutional policies governing human subject research.

2. Will provide subjects with pertinent information (e.g., risks and benefits, contact information for investigators and IRB Chair) and ensures that human subjects will voluntarily consent to participate in the research when appropriate (e.g., surveys, interviews).

3. Assures the subjects will be selected equitably, so that the risks and benefits of the research are justly distributed.

4. Assures that the IRB will be immediately informed of any information or unanticipated problems that may increase the risk to the subjects and cause the category of review to be reclassified to expedited or full board review.

5. Assures that the IRB will be immediately informed of any complaints from subjects regarding their risks and benefits.

6. Assures that confidentiality and privacy of the subjects and the research data will be maintained appropriately to ensure minimal risks to subjects.

7. Will report, by amendment, any changes in the research study that alter the level of risk to subjects.

These criteria are specified in the PI Assurance Statement that must be signed before determination of exempt status will be granted. The PI's signature acknowledges that they understand and accept these conditions. Refer to the Office of Research Support (ORS) website www.utexas.edu/irb for specific information on training, voluntary informed consent, privacy, and how to notify the IRB of unanticipated problems.

1. Closure: Upon completion of the research study, a Closure Report must be submitted to the ORS.

2. Unanticipated Problems: Any unanticipated problems or complaints must be reported to the IRB/ORS immediately. Further information concerning unanticipated problems can be found in the IRB Policies and Procedure Manual.

3. Continuing Review: A Continuing Review Report must be submitted if the study will continue beyond the three year qualifying period.

4. Amendments: Modifications that affect the exempt category or the criteria for exempt determination must be submitted as an amendment. Investigators are strongly encouraged to contact the IRB Program Coordinator(s) to describe any changes prior to submitting an amendment. The IRB Program Coordinator(s) can help investigators determine if a formal amendment is necessary or if the modification does not require a formal amendment process.

If you have any questions contact the ORS by phone at (512) 471-8871 or via e-mail at orsc@uts.cc.utexas.edu.

Sincerely,

James P. Wilson

James P. Wilson, Ph.D.
Institutional Review Board Chair

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